

**Shoal occupancy estimation for 3 lotic crayfish species in the
Tallapoosa River basin, Alabama**

by

Molly Ann Moore Martin

A thesis submitted to the Graduate Faculty of
Auburn University
in partial fulfillment of the
requirements for the Degree of
Master of Science

Auburn, Alabama
May 14, 2010

Keywords: lotic crayfishes, river regulation, adaptive management
occupancy, detection probability, habitat partitioning

Copyright 2010 by Molly Ann Moore Martin

Approved by

Elise Irwin, Chair, Associate Professor of Fisheries and Allied Aquaculture
J. Barry Grand, Professor of Forestry and Wildlife Sciences
James Stoeckel, Associate Professor of Fisheries and Allied Aquaculture

Abstract

The greatest diversity of crayfishes in the world is in the southeastern United States; however many species are at risk and lack of information on habitat requirements and the effects of habitat alteration hamper crayfish conservation efforts (Jones and Bergey 2007, Taylor et al. 2007). Two priority level 2 species (P2; ADCNR) of crayfish are endemic to the piedmont region of the Tallapoosa River Basin; *Cambarus englishi*, and closely related *Cambarus halli*, (Schuster et al. 2008). Additionally, widespread priority level 5 (P5) species, *Procambarus spiculifer*, have been documented in the region (Ratcliffe and DeVries 2004). Conservation of native fauna in large rivers is increasingly dependent on flow management therefore native fauna of the middle Tallapoosa are potentially strongly affected by flow management employed by Harris Dam (Irwin and Freeman 2002).

Occupancy was estimated using methods outlined by Mackenzie et al. 2002 for crayfishes as part of adaptive management of the Tallapoosa River to gain understanding on how flow dynamics affect biota. Specific objectives were to determine variables affecting species specific detection probabilities and compare site level occupancy estimates between regulated and unregulated reaches. Additionally, catch data were examined for differences in size structure among sites. Lotic crayfishes were collected from shoals at 3 regulated and 2 unregulated reaches of the Tallapoosa River basin using pre-positioned area electrofishers (PAE). Detection probability and occupancy were

modeled from presence- absence data as a function of a priori covariates and estimated in Program PRESENCE using the custom single-season single-species models. Model selection was based on the principle of parsimony and superfluous models were eliminated. Weighted model-averaged parameter estimates and unconditional sampling variances were calculated (Burnham and Anderson 2002). Multiple PAE's (i.e. spatial replication; $n= 5-20$) were collected with habitat characters depth, velocity, percent vegetation, and substrate composition recorded and used to model detection. Site level occupancy covariates were based on the a priori hypotheses that occupancy was lower in regulated reaches due to negative impacts of hydropeaking on recruitment and /or occupancy varied along a linear downstream recovery gradient from Harris Dam and one a posteriori hypothesis that occupancy differed among the 5 reaches.

Detection was low for all species in most years which affected precision of occupancy estimates. A few sites consistently had a high number of detections while others consistently had few. Variation in number of detections likely reflected changes in relative underlying populations of crayfishes potentially related to differences in habitat quality, food quality, number of available refuges, or predation risk. At least one individual of *P. spiculifer*, *C. englishi*, and *C. halli* were collected from almost every shoal at least once in the five year sampling period however occupancy estimates varied spatially and temporally. Modeling results suggested occupancy was similar in regulated and unregulated reaches of the basin in a 'wet' year while spatial differences were observed among reaches in all other years. Temporal differences were potentially related to basin hydrology. Data supports occupancy of *P. spiculifer* was close to one ($\Psi \approx 1$) throughout the basin and occupancy of *C. englishi* was higher in the regulated reaches (Ψ

≈ 1) than unregulated reaches ($\Psi \approx 0.50 - 0.60$) in most years. Extremely low detection due to (i.e., sparse data) resulted in model uncertainty making estimates for *C. halli* variable and difficult to interpret. Further investigation of distribution and habitat use for *C. halli* is warranted and *C. halli* may be more abundant in tributaries (Ratcliffe and DeVries 2004). Understanding habitat use of endemic species is important for recommending management actions directed towards conservation of crayfishes.

Habitat covariates supported predicted biological responses, were sensitive to annual basin hydrology, and supported evidence of habitat partitioning among species. Vegetation was important for all species demonstrating a positive effect on detection. Depth influenced detection probabilities in ‘wet’ year and velocity influenced detection in a ‘drought’ year. Catch data also supported evidence of population level responses to drought including changes in size structure and potential density reductions and variation in recovery time among reaches. No evidence supported that the closely related *Cambarus* species competitively exclude one another; however, size differences were observed between species and *C. halli* may limit their use of shoals in the presence of *C. englishi* which may have resulted in consistently low detection of *C. halli* in our study. In addition, depth having a strong influence on detection of *C. halli* and the observed inverse relation to substrate size between the *C. halli* and *C. englishi* may be evidence of habitat partitioning among these closely related species.

Acknowledgements

I would like to thank Dr Elise Irwin and committee for directing my M.S. work and for their encouragement, support, and guidance. I would also like to recognize Dr. Jack Feminella, Dr. Carol Johnston, and the late Dr. George W. Folkerts. Their passions and teachings ignited my interest in aquatic ecology and conservation during my undergraduate studies at Auburn University. Many thanks to Brian Helms and Emily Hartfield for their assistance with crayfish identifications and to all of the Alabama Cooperative Fish and Wildlife Research Unit staff for their field and technical support they provided during this project, as well as the friendship and encouragement. Lastly, I would like to extend special appreciation to my best friend and biggest fan who cheered me all the way through, my husband, Ben.

Table of Contents

Abstract.....	ii
Acknowledgements.....	v
List of Tables	vii
List of Figures	viii
Introduction.....	1
Methods.....	6
Results.....	12
Discussion.....	17
Tables.....	26
Figures.....	35
References.....	49
Appendices.....	55

List of Tables

Table 1.- Water year designation for sampled years in Tallapoosa River basin based on historical hydrologic information	26
Table 2.- Site level CPE's for crayfish species by year and overall rank.....	27
Table 3. Catch data summary for crayfishes by species and by year	28
Table 4.- Covariates used to model detection and occupancy of crayfish species	29
Table 5.- Top detection models for crayfish species by year	29
Table 6.- Occupancy model set for <i>P. spiculifer</i> with K-L final weights.....	30
Table 7.- Occupancy model sets for <i>C. englishi</i> with K-L final weights.....	31
Table 8.- Occupancy model sets for <i>C. halli</i> with K-L final weights.....	32
Table 9.- Annual reach and basin occupancy estimates (Ψ) and naïve occupancy for <i>P. spiculifer</i>	33
Table 10.- Annual reach and basin occupancy estimates (Ψ) and naïve occupancy for <i>C. englishi</i>	33
Table 11.- Annual reach and basin occupancy estimates (Ψ) and naïve occupancy for <i>C. halli</i>	34

List of Figures

Figure 1.- Map of sampling sites in the Tallapoosa River Basin in Randolph, Cleburne, and Tallapoosa counties Alabama	35
Figure 2.- Mean catch-per-effort of crayfishes captured in the Tallapoosa River basin captured between 2005-2009 by season	36
Figure 3.- Mean catch-per-effort of crayfishes captured in the Tallapoosa River basin captured between 2005-2009 by regulation group.....	37
Figure 4.- Carapace length frequency histogram for <i>Procambarus spiculifer</i> captured between 2005-2009 by regulation group	38
Figure 5.- Carapace length frequency histogram for <i>P. spiculifer</i> captured between 2005- 2009 by season.....	38
Figure 6.- Carapace length frequency histogram for <i>Cambarus. englishi</i> captured between 2005-2009 by regulation group.....	39
Figure 7.- Carapace length frequency histogram for <i>C. englishi</i> captured between 2005- 2009 by season.....	39
Figure 8.- Carapace length frequency histogram for <i>C. halli</i> captured between 2005-2009 by regulation group.....	40
Figure 9.- Carapace length frequency histogram for <i>C. halli</i> captured between 2005-2009 by season.....	40

Figure 10.- Carapace length frequency histogram for <i>C. englishi</i> and <i>C. halli</i> captured between 2005-2009.....	41
Figure 11.- Average annual detection probability values by species.....	41
Figure 12.- Relation between detection and depth for 3 levels of vegetation <i>P. spiculifer</i> and <i>C. halli</i>	42
Figure 13.- Relation between detection and velocity with the influence of substrate size for <i>P. spiculifer</i>	43
Figure 14.- Relation between detection and velocity for 3 levels of vegetation for <i>P.</i> <i>spiculifer</i>	43
Figure 15.- Relation between detection, velocity, and depth for <i>C. halli</i>	44
Figure 16.- Relation between detection and percent vegetation with the influence of substrate size and velocity for <i>C. englishi</i> and unidentified YOY.....	45
Figure 17.- Relation between detection and vegetation with the influence of substrate size for <i>C. englishi</i>	46
Figure 18.- Relation between detection and depth with the influence of vegetation and substrate size for <i>C. halli</i> and <i>C. englishi</i>	47
Figure 19.- Model averaged estimates of reach scale occupancy by year and species.....	48

INTRODUCTION

The greatest diversity of crayfishes in the world is in the southeastern United States where more than 300 species of approximately 540 species worldwide are extant (Taylor 2002). However, between 1/3 and 1/2 of crayfish species are at risk of serious decline or even extinction (Taylor and Schuster 2004). Narrow geographical ranges and limited distributions increase susceptibility of many southeastern species to extinction. Furthermore, lack of information on habitat requirements and the effects of habitat alteration hamper crayfish conservation efforts (Taylor 2002, Jones and Bergey 2007). Effective conservation of these species will require life history and distribution information which is available on less than 40% on North American crayfishes (Taylor et al. 2007). Two priority level 2 species (P2; ADCNR) of crayfish are endemic to the piedmont region of the Tallapoosa River Basin; *Cambarus englishi*, and closely related *Cambarus halli*, (Schuster et al. 2008). Additionally, 2 widespread priority level 5 (P5) species, *Procambarus spiculifer*, and *Cambarus latimanus*, have been documented in the region (Ratcliffe and DeVries 2004).

Crayfishes are important members of lotic communities influencing ecosystem level processes of energy flow and nutrient transfer. When present, crayfish typically represent the largest biomass of macroinvertebrates (Hury and Wallace 1987, Rabeni et al. 1995) and may directly or indirectly influence populations at multiple trophic levels. Multiple species of predatory fishes, birds, and mammals feed on crayfishes and trophic

complexity is increased by the opportunistic feeding of crayfishes on macrophytes, algae, detritus, macroinvertebrates, amphibians, and fish (Mormot 1995). Molting crayfish are vulnerable to both intra- and inter-specific attacks and adults frequently cannibalize juveniles, especially after the maternal pheromone ceases (Rabeni 1985). Crayfishes perform important ecosystem functions such as influencing sediment movement (Helms and Creed 2005) and processing macrophytes and leaf litter into fine particulate organic matter which may be used by other stream organisms (Huryn and Wallace 1987, Whitley and Rabeni 1997). Additionally, sensitivity to common classes of pesticides, organophosphates and carbamates, make crayfish useful bioindicators (Hyne and Maher 2003).

Occupancy, the probability of species occurrence, is a measure used to quantify the status of a population or community. Changes in occupancy may therefore serve as a basis for conservation and management decisions (Nichols et al. 1998, Peterson and Dunham 2003, Gu and Swihart 2004, MacKenzie et al. 2006). Until recently occupancy estimation procedures did not take into consideration false absence of species or individuals during surveys (Yoccoz et al. 2001). Mackenzie et al. (2002) developed modeling techniques using maximum likelihood to estimate occupancy and when detection probabilities are less than 1. Species detection probability is defined as the probability of detecting at least one individual of the species during sampling of an occupied site (MacKenzie et al. 2006). Detection is most often both species and survey specific and therefore may vary spatially and temporally depending on various factors such as sampling method, habitat, observer, local extinction and colonization rates, underlying population size or density, and seasonal behaviors (Mackenzie et al. 2002,

Royle and Nichols 2003, Bailey et al. 2004, Hayer and Irwin 2008). Recognizing that non-detections do not always imply species absence, it is necessary to model variability in detection probabilities (p) to obtain an unbiased estimate of occupancy (Ψ). Therefore, simultaneous modeling of detection and occupancy reduces bias by estimating the probability of absences and failure to detect a species when present.

Conservation of native fauna in large river systems is increasingly dependent on flow management. Hydropeaking activities decrease persistence of shallow habitats which facilitate reproduction and YOY (young-of year) survival of various taxa (Freeman et al. 2001). Therefore, native fauna of the middle Tallapoosa are potentially strongly affected by flow management employed by R. L. Harris Dam. We modeled detection and occupancy of crayfishes to help inform adaptive management for the Tallapoosa River below R. L. Harris Dam which began in 2005. Adaptive management (Walters 1986) is the evaluation of system response to management with emphasis on reduction of uncertainty to guide further prescribed management. One of the primary objectives for adaptive management of the Tallapoosa River is to determine faunal response to various flow management prescriptions. Occupancy was estimated for various taxa, including crayfishes, to gain understanding on how flow dynamics affect biota.

Objectives of this study were to:

- 1) Estimate occupancy of crayfish by species in regulated and unregulated river reaches.
- 2) Determine variables affecting species-specific detection probabilities.
- 3) Compare crayfish community and size structure in regulated and unregulated reaches of the Tallapoosa River basin.

Crayfishes exhibit spatial heterogeneity in their distributions due to habitat preferences (Maude and Williams 1983, Mitchell and Smock 1991, Flinders and Magoulick 2007) and anti-predatory behavior (Stein and Mangson 1976, Englund and Krupa 2000, Fortino and Creed 2007, Olsson and Nystrom 2009). Often closely related species will exhibit disjunct distributions perceived to be a result of competitive exclusion given the aggressive nature of crayfishes, accordingly species specific habitat partitioning and ontogenetic shifts in habitat have been observed in crayfishes (Rabeni 1985, Flinders and Magoulick 2007). Crayfish distribution is often related to size specific predation risk where small crayfish prefer shallow habitat with gravel or small cobble substrates because the depth and interstitial space prevents predation from fish and larger crayfish; whereas, larger crayfish prefer slower moving, deeper water to reduce predation risk from terrestrial predators (Stein and Mangson 1976, Maude and Williams 1983, Mitchell and Smock 1991, Kershner and Lodge 1995 Englund and Krupa 2000, Flinders and Magoulick 2007, Fortino and Creed 2007). Additionally, demographics such as body size distribution may correspond to population level processes through contribution to survival and fecundity (Peters 1983, Werner and Gilliam 1984).

Species detectability is often related to the number of individuals at a site or within a sampling unit which is likely to be affected by behavior and habitat use or availability. Season, temperature, and light conditions influence behaviors including foraging activity, reproduction, and molting cycles (Hobbs 1942, Gore and Bryant 1990, Bubb et al. 2004). The relation between detection probability and stream habitat features may provide inference on habitat requirements (Distephano et al. 2003a, Flinders and Magoulick 2007). The combination of velocity and depth control the distribution of

substrate particles and food and therefore influence the distribution of crayfish (Gore and Bryant 1990). However, amount of cover and behavioral patterns were expected to have the greatest influence detectability. Specifically, detectability was predicted to exhibit a positive relation to percent vegetation and velocity because more abundant vegetative cover may increase the number of individuals in the sampling unit and swift current will aid in our capture method (see field methods). Detectability was predicted to increase in low light conditions due to increases in crayfish activity and movement. Variable effects of substrate type and depth among species and between adults to juveniles were expected and may reflect habitat preferences. Occupancy estimates were predicted to be lower in regulated reaches due to negative effects of hydropeaking activities on recruitment (Freeman et al. 2001), and potentially increase along a downstream recovery gradient from the dam (Bain et al. 1988, Kinsolving and Bain 1993, Travnicek et al. 1995).

METHODS

Study Sites

All sampling sites were located in the piedmont physiographic province of the Tallapoosa River Basin in East Alabama, USA. The study area contains extensive shoals, shallow river habitat features that characteristically support high faunal diversity (Irwin and Freeman 2002). Sample stratification by habitat is often used to sample aquatic biota that exhibit heterogeneous or clustered distributions (Distefano et al 2003b). Shoals are ideal study areas in that within a shoal, various microhabitat types (i.e., riffles, runs, and shallow pools) are represented thus random sampling of randomly selected shoals will incorporate the variety of available habitats allowing for broader inference in the system.

A probabilistic sampling approach was employed where 5 shoals were randomly selected from within 5 river reaches that varied in length (Figure 1); 3 regulated reaches and 2 unregulated reaches for a total of 25 shoals from 5 reaches (Irwin and Freeman 2002). The regulated segment beginning at Harris Dam and terminating downstream in the headwaters of Martin Reservoir was divided into 3 reaches; Dam to Malone (0-11 km from dam), Malone to Wadley (11-22 km from dam), and Griffin Shoals to Jaybird Creek, also known as Horseshoe Bend (60- 90 km from dam). Two unregulated reaches; Hillabee Creek between Sanford Rd and Hwy 22, and the Tallapoosa River above R.L. Harris Dam between Ben Mills and Evans Road were monitored to assess how occupancy varied independent of regulated flows. USGS gages were located near all

sites. Annual mean discharge data from the period of record available for each gage was used to designate water year classification (Table 1).

Field Methods

Crayfish species in temperate regions are mostly active from April- November, typically breeding in late fall or early spring with females carrying attached young during April and the beginning of May (Taylor and Schuster 2004). Sites were visited 2 times per year, once in the summer (late May – early August) and once in the fall (mid-September- November). Crayfishes can be difficult to sample and electrofishing is regarded as an effective method for collecting lotic crayfishes as other methods such as trapping typically demonstrate sex and size bias, require more visits and potentially gain less information (Rabeni et al.1997). However, unlike most fishes, crayfishes are not immobilized from the current and respond with rapid, convulsive backward movement (Minckley and Craddock 1961) which may have implications about our inability to detect crayfishes even when present in a sampling unit. Crayfishes were collected using 9m² (1.5m X 6m) pre-positioned area electrofishers (PAE). PAE's were left in place undisturbed for at least 10 minutes and then electrified using alternating current AC current for 20 seconds (Honda 2.5 GPP; Type VI-A Electrofisher; Smith Root[®], Inc., Vancouver, Washington) while 2 observers each held one end of a seine along the downstream border of the PAE and a dip net to capture specimens. Afterward a third observer captured any visible specimen(s) and then disturbed the substrate dislodging any additional specimens to be collected in the downstream seine.

Based on previous research (Freeman et al. 2001), 2005-2007 protocol was to collect 20 samples per visit and in 2008-2009 effort was reduced to 10 samples based on

preliminary analysis of fish data (Irwin et al. 2009). Occasionally reduced wetted area due to low flows prohibited 10-20 samples and in these cases as many samples as wetted channel would permit were collected. Additionally no samples were obtained from certain sites in whole seasons due to site accessibility circumstances in year of drought or because of depth related sampling gear limitations in wet years (see Appendix 1 for sampling dates). Specimens from 2005-2007 were preserved and brought back to the lab for processing and 2008-2009 samples were field identified, measured, and released. Carapace length (CL = tip of rostrum to post-median margin of carapace) was measured to the nearest 1.0 mm using calipers. Specimens were sexed and identified to species when possible (Hobbs 1981); individuals smaller than 14 mm in carapace length were not identified to species and were classified as YOY (young-of-year). Due to erroneous field identification during the summer of 2008, no specimens were identified to species, and were simply classified as either adult or YOY based on aforementioned carapace length criteria.

Specific microhabitat features were measured for each PAE sample. Features recorded were: depth, velocity, percent vegetation, and substrate composition. Depth and velocity were measured using a Marsh-McBirney flow staff and meter. Substrate composition and areal vegetative cover and type were quantified by visual estimation. Substrate particle designation were recorded in the order of dominance and defined as silt (<0.1 mm), sand (0.1-1 mm), gravel (0.1 -6 cm), cobble (6 – 12 cm), boulder (>12 cm), continuous bedrock, bedrock ledge, and small (1 cm - 4 cm diameter) or large (>4 cm diameter) woody debris. Additional variables recorded for each PAE were date, time, water temperature, and weather conditions (i.e., sunny, partly cloudy, or overcast).

Data Analysis

Catch data were examined for differences between seasons and regulated versus unregulated sites in CPE (catch-per effort; crayfish/PAE) using Kruskal Wallis tests and in carapace length distributions using Kolmogorov - Smirnov tests (SAS v9.1/ SAS Institute Inc., Cary, NC). Detection probabilities and occupancy were estimated using maximum likelihood methods modeled as a function of covariates using the logit link function using single-season, single-species models in Program PRESENCE v. 2.2 (<http://www.mbr-pwrc.usgs.gov/software/presence.html>, Hines 2006) following the approach of MacKenzie et al. 2006. To collect occupancy data multiple PAE samples ($n = 5-20$; see Field Methods) were taken at each shoal visit; each PAE sample was considered a sampling occasion (i.e., spatial replication). Each year was considered a season and each shoal considered a site. Occupancy modeling incorporates 2 types of covariates; site specific covariates that effect occupancy estimation (Ψ) and survey specific covariates that effect detection probability estimation (p). Detection and occupancy covariates modeled were based on the a priori hypotheses that detection probabilities varied by environmental and habitat characteristics sampled within each PAE and that occupancy differed between regulated and unregulated segments of the Tallapoosa, and/or along a linear gradient downstream from Harris Dam. Environmental and microhabitat features of each sampling occasion (PAE) were used as covariates in modeling detection probabilities. All variables could be modeled as a continuous numerical value except light conditions and substrate which were transformed into a categorical value. Light conditions were expressed binomially (0, high light and 1, low light; i.e., cloudy or overcast). Substrate particles were converted to a categorical number

using values modified from Gore and Bryant 1990, values reflecting the capacity of the substrate to provide refuge and to alter micro velocities; bedrock and silt = 0, sand = 1, gravel = 2, cobble = 3, small woody debris = 3.5 bedrock shelf = 4, large woody debris = 4.5, boulder = 5. Gore and Bryant used these values to calculate a roughness index, however to allow more straightforward interpretation we did not calculate roughness but used only the largest substrate present in the PAE. Substrate observations were made at the 9m² scale which was coarse in relation to the size of crayfishes, therefore using only the dominant substrate was not considered the most biologically relevant way to represent the influence of substrate.

Model selection criteria were based on the principle of parsimony; competing models were compared using Akaike's information criterion (AIC; Burnham and Anderson 2002). To limit the number of models in each model set, a 2-step approach was employed to analyze data. First, detection trials were run to determine which variable(s) best explained detection probabilities. Then top detection models were then combined with site level covariates which represented hypothesized differences in occupancy between sites to create a final candidate model set. Detection and final model sets were examined for superfluous covariates (i.e., covariates that did not improve model fit; Burnham and Anderson 2002). Final candidate models with covariates that did not add substantial model support were eliminated and model weights were re-calculated. Inference was based on a single 'best' model if the top model weight ≥ 95 ; otherwise, weighted model-averaged parameter estimates and unconditional sampling variances were calculated on the Kuller- Leiback (K-L) confidence set for the best model created by simply taking the sum of model weights from largest to smallest until the sum reached

≥ 95 (Burnham and Anderson 2002). Each model set was assessed for lack-of-fit using the most global model, by calculating Pearson chi-square statistic

Equation 1.

$$X^2 = \sum_{i=1}^n \frac{(O_i - E_i)^2}{E_i}$$

where O_i and E_i are the observed and expected numbers of observations for class i , and n is the total number of classes defined by the current model, for 100 parametric bootstraps to determine if the observed statistic for the data set was unusually large (MacKenzie and Bailey 2004). Substantial lack-of-fit may lead to erroneous inferences resulting from error in bias or precision of parameter estimates and their associated standard errors.

Variance inflation factor, \hat{c} , values were used to estimate to identify overdispersion by dividing the chi-squared statistic by the average observed chi-squared test statistic from the bootstraps. Overdispersion resulted from small violations in assumptions however, large \hat{c} values (>4) suggest inappropriate model structure (Burnham and Anderson 2002). Overdispersion does not usually increase bias of parameter estimates but may underestimate error. Therefore, quasi-likelihood analysis criteria (QAIC) were used to rank any model sets with values of $\hat{c} > 1$.

Equation 2.

$$QAIC = - \left[\frac{2 \text{Log} \left(L(\hat{\theta}) \right)}{\hat{c}} \right] + 2K$$

RESULTS

Catch Data and Size Structure

CPEs (catch-per-effort) at individual sites were highly variable ranging from 0 to 3.05 crayfish/PAE. Overall CPEs were lowest in summer 2007 and summer 2008 (Figure 2). Table 2 reports annual site level CPEs and ranks sites over the 5 year period with catch rates at sites within Hillabee Creek and Horseshoe Bend reaches dropping dramatically after 2006. Catch data supported that YOY crayfish did not recruit to sampling gear until reaching the size of approximately 8 mm; catch data from all years of suggests that this occurs around the second week of June in both regulated and unregulated reaches. In most years, regulated sites were sampled before June 1st and when YOY were excluded and all years were pooled no significant differences ($X^2 = 0.15$, $DF=1$; $p=0.47$) were observed between CPE at regulated versus unregulated sites (Figure 3). However, in 2009 regulated sites (0.70 crayfish/PAE; CI: 0.40-0.99) had higher ($X^2 = 3.6$, $DF=$; $p=0.06$;) mean CPEs than unregulated sites (0.22 crayfish/PAE; CI: 0.01-0.43) YOY excluded.

Over the five year sampling period a total of 1650 crayfish including YOY were sampled using PAE's (Table 3). Species identified were: *P. spiculifer* ($n=572$), *C. englishi* ($n=299$), and *C. halli* ($n=151$). When all data were pooled, *P. spiculifer* individuals captured at regulated sites (32.1mm; CI: 31.1mm-33.1mm) were significantly larger ($p < 0.0001$) than those from unregulated (27.6 mm; CI: 26.5mm-29.1mm; Figure 4) and individuals captured in the summer (34.5 mm; CI: 33.5.1mm-35.5mm) were

significantly ($p < 0.0001$) larger than those captured in the fall (26.2 mm; CI: 25.1mm-27.2 mm; Figure 5). When analyzed by year and season, carapace lengths were significantly different ($p < .0001$) between regulated (36.2 mm; CI: 34.8 mm – 37.7 mm) and unregulated (16.8 mm; CI: 15.5 mm – 18.3 mm) sites in the summer of 2007. Additionally, mean body size of *P. spiculifer* decreased significantly ($p < 0.001$) between 2006 (26.2 mm; CI: 23.8 mm – 28.5 mm) and 2007 (17.7 mm; CI: 16.0 mm - 19.4 mm) at unregulated sites; whereas, mean body size did not significantly differ between years at regulated sites ($p = 0.93$).

Carapace lengths of *C. englishi* captured in regulated reaches (26.3 mm; CI: 25.8 – 26.9 mm) were significantly larger than individuals captured in unregulated reaches (22.1 mm; CI: 19.8 mm -24.5 mm) when all data were pooled (Figure 6**Error! Reference source not found.**). *C. englishi* captured in the summer (27.2 mm; CI: 26.7 mm – 27.7 mm) were significantly ($p < .0001$) larger than those captured in the fall (23.7 mm; CI: 22.6 mm-24.8 mm; Figure 7). When examined by season and year, significant differences ($p < .0001$) in carapace length of *C. englishi* between regulated (27.5 mm; CI: 26.4 mm – 28.6 mm) and unregulated (16.2 mm; CI: 14.4 mm – 17.9 mm) were found in the summer of 2007. When all *C. halli* data were pooled, significant differences ($p < 0.0001$) were observed in carapace length distributions between regulated (23.4 mm; CI: 22.1 mm-24.7 mm) and unregulated (19.2 mm; CI: 18.2 mm - 20.2 mm; Figure 8) sites. However, no seasonal differences ($p = 0.14$) were observed; therefore seasonal data were pooled for annual comparisons (Figure 9). In 2007, carapace lengths of *C. halli* differed ($p < 0.0001$) between regulated (24.6 mm; CI: 22.0 mm – 27.2 mm) and unregulated sites (16.8 mm; CI: 15.6 mm – 17.9 mm). Additionally, carapace lengths of the 2 *Cambarus*

species were significantly different ($p < 0.001$) where captured specimens of *C. englishi* (25.9 mm; CI: 25.3 mm – 26.4 mm) were on average larger than *C. halli* (21.3 mm; CI: 20.4 mm- 22.2 mm; Figure 10).

Detection Probabilities and Occupancy

Average detection probabilities were low for all species in most years ($p < 0.15$). Detection remained fairly constant across years for *C. halli* ($p = 0.03 - 0.07$); however, in 2008 and 2009 detection probabilities for *C. englishi* and *P. spiculifer* increased compared to the other years (Figure 11). Detection probabilities were a function of habitat variables for all species; habitat covariate values and detection histories are summarized in Appendices 2-8. Preliminary model results suggested environmental variables date, temperature, and weather were poor predictors. For example, low light conditions which were hypothesized to increase detection for all species of crayfishes by increasing crayfish activity but exhibited a positive relation for one species and a negative for another or for the same species response would alternate between positive and negative relation between years. Therefore, environmental covariates were eliminated from further modeling exercise. Furthermore, dramatic decline of catch rates in Hillabee Creek and Horseshoe Bend reaches provided the impetus for the a posteriori site (Ψ) level covariate ‘reach’ to separate these from their respective regulation groups (Table 4).

Table 5 indicates the top detection model for each model set and Figures 12-18 demonstrate the relation of species detection to specific habitat characters. Vegetation positively influenced detection and was identified as an important covariate explaining detection for all model sets except for *P. spiculifer* 2008 detection models. When

identified as an important covariate, depth consistently had a negative effect on detection for all species; whereas, the magnitude of the effect differed among species and years (Figure 12). Depth affected detection of all species in 2005. Velocity had variable effects on *P. spiculifer* influencing detection positively in 2008 and negatively in 2009 (Figure 13-14) and demonstrated a positive influence on detection for both *Cambarus* sp. And YOY when identified in top detection models (Figure 15-16). Relation and relative importance of substrate size varied among species (Figures 16-18). Every year, substrate size was identified as a top covariate positively influencing detection of *C. englishi*. When identified in top models substrate size positively influenced detection of *P. spiculifer*, but negatively influenced detection of YOY and of *C. halli* when identified in top models Additionally, due to known bias in the data of sampling date on YOY capture histories (i.e., recruitment) YOY data was used to model detection only.

Three data sets were identified as overdispersed; *P. spiculifer* 2009, *C. englishi* 2005 and 2009 with variance inflation factor c -hat values of 1.1, 1.2, and 1.3 respectively; therefore QAIC was used to rank these models. Data suggested occupancy was similar throughout the basin for all species in 2005, whereas potential differences among reaches were supported for most species in all other years (Table 6- 8). Annual basin and reach scale occupancy estimates and standard errors are presented by species in Table 9-11 and Figure 19.

Model certainty was high for *P. spiculifer* occupancy models allowing inference in most years to be based on one model. Estimates for *P. spiculifer* indicate occupancy was close to one ($\Psi \approx 1$) and fairly stable in the Upper Tallapoosa River and the 2 regulated reaches closest to R. L. Harris dam (i.e., Dam to Malone and Malone to

Wadley) although a decrease was observed in the 2 regulated reach estimates in 2008. Modeling results suggest that occupancy of *C. englishi* was higher in regulated reaches ($\Psi \approx 1$) with a slight negative effect of distance from R.L. Harris dam, compared to unregulated reaches ($\Psi \approx 0.50$). However in 2008, estimates were around 60% ($\Psi \approx 0.60$) for all reaches. In the Upper Tallapoosa reaches occupancy estimates for *C. englishi* were fairly stable ($\Psi = 0.55-0.65$), but imprecise and variable in Hillabee Creek. Estimates for *C. halli* did not support any consistent pattern of occupancy across sites and years; however naïve occupancy and occupancy estimates declined over the sampling period in Hillabee Creek and exhibited the least variability in the Dam to Malone reach.

DISCUSSION

Data and modeling results suggested that occupancy varied by species, reach and in relation to river regulation and distance from R. L. Harris dam. Detection probabilities also varied by species and habitat. In addition to providing unbiased estimates of occupancy, detection probabilities were useful in definition of habitat variables that may be important to distribution of these species. In addition, length frequency analysis was useful in identification of recruitment events, species specific population characters and differences in size structure between populations in regulated and unregulated reaches of the Tallapoosa basin. Of particular interest to conservation managers was the finding that occupancy estimates reflected conservation status where the P5 species (of least concern), *P. spiculifer*, had highest occupancy throughout the basin and was stable across years; whereas, data suggested both P2 species (GCN) *C. englishi* and *C. halli* may have more limited distributions. In relation to adaptive management of flow regimes below the dam, findings will be used to assist managers with decisions relative to future modifications of the flow regime. The unexpected finding that *C. englishi* had higher occupancy below the dam indicated that relations among hydrology, habitat, and occupancy need to be defined.

In general, the hypotheses related to occupancy for the 3 species were supported by the data. Modeling results indicated temporal and spatial variation in occupancy. Occupancy estimates were similar in similar hydrological years with higher occupancy in

normal-dry years, 2006 and 2009, than in drought years, 2007 and 2008. In 2005, a ‘wet’ year, occupancy was constant throughout the basin for all species whereas in other years spatial variation was exhibited for all three species. In 2007-2009 ‘reach’ explained the most variation in occupancy for *P. spiculifer* and *C. halli* whereas the 2 regulation related hypotheses best explained variation in data for *C. englishi* in most years with higher occupancy in regulated reaches. Modeling only fall data in 2008 may have contributed to reduced naïve occupancy for all species compared to other years as a portion of the population may have been unavailable to sampling due to seasonal behaviors. In 2009, *P. spiculifer* and *C. englishi* data sets were identified overdispersed; the overdispersion resulted from unmodeled differences in detection between shoals, with 3 shoals in particular having extremely high detections relative to other sites and years (Malone B, Malone E and Wadley C) potentially indicating increased local population size.

Occupancy of *P. spiculifer* in the basin was likely constant and close to 1. Although, in 2007-2009 ‘reach’ explained the most variation in occupancy for *P. spiculifer* and in those years Hillabee Creek and Horseshoe Bend had variable and imprecise estimates due to very low number of detections. Variations in estimates likely resulted from differences related to habitat or fluctuations in population size. In the summer of 2009, at Hillabee B no crayfish were detected in PAE samples, but 1 *P. spiculifer* was collected in a macroinvertebrate Surber sample and additionally, crayfish were collected using a backpack electrofishing unit in deeper habitat at Horseshoe Bend reaches in 2008 and 2009 (Irwin, unpublished data). During 2007-2009, use of shoal habitats by *P. spiculifer* and other crayfishes may have differed and/or smaller underlying

populations may have been present. These differences were potentially in response to biotic or abiotic variability and any apparent gradient in occupancy may reflect a gradient in abundance (MacKenzie et al. 2006). Existing abundance models for occupancy (Royle et al. 2005) currently are unable to account for temporal variability in the amount of aquatic habitat, because changes in flow determine the amount and availability of specific habitats. In addition, relation between habitat amount, persistence and/or stability and crayfish occupancy have not been quantified.

Similarly to *P. spiculifer*, changes in occupancy estimates for *C. englishi* may reflect changes in population size or shoal use where *C. englishi* demonstrated higher occupancy (i.e. shoal use or density) in years with ‘normal’ mean discharge. Data indicated that *C. englishi* have higher occupancy rates and were more widespread in the regulated reaches. Naïve estimates and occupancy estimates in the regulated reaches were similar in similar hydrologic years; highest in ‘normal- dry’ years and decreasing in years of ‘drought’, but lowest in a ‘wet’ year. In the ‘normal’ years occupancy is estimated to be close to 1 in the regulated reaches. Decreased occupancy in regulated reaches in 2005 ‘wet’ years versus other years ($\Psi= 0.68$) potentially indicated effects of increased frequency hydropower generation. Additionally, model uncertainty was high in most years for *C. englishi*. The inability to determine a best model suggests that the data were inadequate to make strong inferences potentially due to some ambiguous effect of parameterization or structure (Burnham and Anderson 2002). Fortunately, information theoretic approach allows formal multi-model inference in the face of model uncertainty. Therefore, model averaged parameter estimates based on the observed data were not conditional to the single observed data set.

Despite having the lowest detection probability, model uncertainty was low for *C. halli* in 2005-2007, but high in 2008-2009. Estimates for *C. halli* were difficult to interpret; however naïve occupancy declined throughout the sampling period whereas detection remained relatively constant. Low naïve occupancy in 2008-2009 was potentially related to reduced sampling effort for a rare species. Furthermore, the probability that an occupied site goes undetected with detection probability $p = 0.03$ is $(1-p)^s$ ($s = \#$ of surveys), 54% for 20 sampling occasions and 74% for 10 sampling occasion respectively. With p this low, models are unable to resolve which sites were “unoccupied” and which sites were occupied but not detected. Although catch of *C. englishi* was higher, occupancy estimates were similar and often higher for *C. halli* than *C. englishi* emphasizing the importance of modeling exercises to avoid bias associated with raw catch data. To make the best use of data for a species with extremely low detection such as *C. halli* ($p = 0.03 - 0.07$), a suggested method is to pool data with a similar species that has a similar detection probability (Mackenzie et al. 2006). However, detection model results suggested that it would be inappropriate to pool *C. halli* with any of the other species as detection probabilities were influenced by different covariates. Additionally, basin occupancy estimates (i.e., all sites) for *C. halli* steadily decreased during the sampling period although detection models and other data support this could be the result of limited use of shoal habitat.

Habitat covariates were useful in modeling crayfish detection. The relation of species specific detection to stream habitat features corroborated predicted biological relations and provided insight on habitat use, supporting evidence of habitat partitioning among species. Furthermore, results were sensitive to annual variation in basin

hydrology and catch data and demographics supported modeling results. Detection models supported predicted biological relations to habitat characters and differences in species specific relation to habitat characters indicated differential habitat use among species. Vegetation had a positive influence on crayfish detection probabilities in all but one detection model which supported the importance of vegetation as refuge (Rabeni 1985, Mormot 1995, Distephano et al. 2003a, Flinders and Magoulick 2007, Brewer et al. 2009). Models indicated depth and vegetation were the most important variables affecting detection of *P. spiculifer* in 2005-2007; whereas, in 2008 velocity and substrate were the most important variables. Only fall data was modeled in 2008 and the difference may be the result of habitat responses related to seasonal changes in flow and vegetation cover in fall months (i.e., lower flows, less vegetation). However, no change in detection covariates was observed for the 2 *Cambarus* species. Additionally, in 2009 top detection covariates for *P. spiculifer* were velocity and vegetation and detection demonstrated a negative relation to velocity. Inconsistent selection of important detection covariates for *P. spiculifer* could be the result of a generalist habitat use.

Detection of YOY crayfishes was positively related to vegetation and negatively related to substrate size supporting their preference for gravel substrates because the interstitial refuge provided prevents predation from fish and larger crayfish (Stein and Mangson 1976, Flinders and Magoulick 2007, Ollsson and Nystrom, 2009). Detection probabilities of *C. englishi* were influenced by vegetation and substrate every year and consistently demonstrated a positive relation to substrate with higher detection over larger substrates such as boulders which were common in the 2 upper regulated reaches where catch rates for *C. englishi* were consistently highest. Furthermore, *C. englishi*

were detected more frequently at Upper Tallapoosa sites composed mainly of large gravel substrates than Hillabee Creek which shares similar bedrock and boulder substrate features of shoals in the mainstem Tallapoosa River. Therefore, data suggested that presence of boulder substrate habitat does not influence presence or occupancy of *C. englishi*, but when present, boulders and other large substrates types may be important habitat features influencing *C. englishi* distribution in a lotic system.

The presence of a particular species may affect detection or occupancy of another species (MacKenzie et al. 2006). Closely related crayfish species often exhibit allopatry perceived to result from overlap in resource use, however, *C. englishi* and *C. halli* are often sympatric (Bouchard 1978, Hobbs 1981, Ratcliffe and DeVries 2004). Although differences in carapace lengths reflected in our data suggested that *C. englishi* were on average larger than *C. halli* there was only one shoal where *C. halli* was never found in the presence of *C. englishi*; thus there is little evidence to support that either *Cambarus* species competitively excluded one another. However, *C. halli* may limit their use of shoals in the presence of *C. englishi* which may have resulted in lower detection of *C. halli* in our study. Depth having a strong influence on detection of *C. halli* and the inverse relation to substrate size between the *C. halli* and *C. englishi* may be evidence of habitat partitioning among these closely related species.

Our findings were supported by a recent study in of the 2 species in the Little Tallapoosa River basin in Georgia reporting that at sympatric sites *C. halli* was smaller than *C. englishi* and shifted habitat use to exploit shallow riffles (Dennard et al. 2009). Hobbs (1981) reported the contrary, finding when both species were present *C. halli* did not use riffles but exploited pools. Larger, ‘adult’ individuals of *C. halli* may use pool

habitat > 1 m in depth which we did not sample this would support observed differences in size structure. Observed size differences among the 2 species may also be the result of asynchronicity in the release of YOYs which has resulted in differing size structures of 2 similar *Orconectes* spp. (*Orconectes luteus* and *Orconectes punctimanus*) in Ozark streams (Rabeni 1985). However, we did not identify YOY to species, but both of these factors may have contributed to catch and size differences among the two species.

Crayfish carapace lengths are indicators of population level demographics and processes and can reflect responses to both biotic processes and abiotic variability. Seasonal differences in carapace length distributions were the result of a portion of the population being 'unavailable' for sampling in fall due to behaviors associated with molting and/or reproduction and from the surge of summer recruits transitioning to small adults (Taylor and Shuster 2004). In response to drought, Taylor (1988) observed smaller mean crayfish body size, lower overall density, reduced abundance, and an increase in the proportion of YOYs. Extremely low water levels during the sampling period of 2007 in unregulated reaches likely had an effect on crayfish body size as significantly smaller carapace lengths were observed at unregulated sites for all species in 2007. Smaller size was likely maintained through 2008, but data did not allow these comparisons due to lack of species specific information in the summer and small to no sample size for some species in the fall. However, in the summer of 2008 the largest 'adult' captured in the Upper Tallapoosa (n=40) was only 18 mm. This difference was not observed at regulated sites likely because low water levels are mitigated during drought by conservation flow management from R.L. Harris Dam.

Increased detection probability in 2008 and 2009 may have partially resulted from a new observer in 2009, but likely also reflect changes in abundance in response to drought. (Taylor 1988). The proportion of PAEs where *C. englishi* was detected in 2008 and 2009 was 2 times higher than in 2005-2007 and for *P. spiculifer* detection were similar to previous years in 2008 but on average about 40% higher in 2009 than other years, increasing particularly in regulated reaches in 2009. An increase in detection was not observed for *C. halli*, therefore low but consistent detection of *C. halli* in the sampled shoal habitat suggested differential and perhaps limited use of shoal habitat for *C. halli*. Furthermore, large proportions of YOYs at unregulated sites in 2009 indicated the sites may still have been in recovery from drought-induced population effects. In 2009, increased occupancy and particularly high numbers of detections of adults at some regulated sites, potentially resulted from enhanced recruitment afforded from reduced hydropeaking activities in previous years. Additionally, in 2008 there were a high number of detections for YOY at regulated sites. Freeman et al. (2001) reported that recruitment events in fishes were strongly related to habitat persistence, particularly in terms of long periods of non-generation below R.L. Harris dam.

Furthermore, detection models supported that variation in annual hydrology affected crayfish distributions on shoals or microdistribution. Depth influenced detection probabilities for all species in 2005 which was considered a ‘wet’ year based on historical means for the system with a greater proportion of sampling units were in depths > 61 cm in 2005 than other years. Velocity positively influenced detection of YOY and both *Cambarus* species in 2007, classified a ‘drought’ year and having a greater proportion of sample units with velocities between 0-10 cm/s than other years. Velocity was also

identified in the top detection model for *C. halli* in 2006 potentially indicating higher sensitivity to changes in velocities as 2006 was considered a normal to dry year.

Variation in hydrology may have affected crayfish microdistribution; therefore, flow management has the potential to affect crayfish microdistribution ultimately influencing metapopulation processes of extinction or colonization. Furthermore, variation in annual occupancy estimates may also be potentially related to differing hydrology. Multi-season models using water-year as a covariate to explain extinction/colonization rates could be developed to test this hypothesis.

Our data demonstrated that flow management implemented by R.L. Harris dam has the potential to effect crayfish body size, microdistributions, recruitment, and changes in species occupancy. Occupancy estimates interpreted along with detection results provided quantitative information useful for evaluating population status and distributional patterns of crayfishes in the Tallapoosa River basin. Occupancy estimates for P5 species *P. spiculifer* was constant and essentially 100%. Furthermore, occupancy estimates for the P2 species were lower although occupancy of *C. englishi* was close to 1 in regulated reaches in 'normal' years. Further investigation of distribution and habitat use for *C. halli* is warranted and *C. halli* may be more abundant in tributaries (Ratcliffe and DeVries 2004) or may exhibit different use of habitat in mainstem versus tributaries. Understanding habitat use of endemic species is important for recommending management actions directed towards conservation of crayfishes.

Table 1.- Water year designation for sampled years in Tallapoosa River basin based on historical hydrologic information. Inflow to R.L Harris reservoir includes Upper Tallapoosa and Little Upper Tallapoosa gages to reflect basin hydrology.

Water Year	Designation	Inflow R.L. Harris ^a	Hillabee	Classes
2005	Wet	1602.3	422.7	
2006	Normal - Dry	967.3	175.7	
2007	Drought	476.8	130	
2008	Drought	442.1	132.3	
2009 ^b	Dry-Normal	883.8	275.3	
		> 1779	> 423	Flood
		1556-1779	363-422	Wet
		914-1556	230-362	Normal
		619-914	133-229	Dry
		< 619	< 133	Drought

^a USGS 02412000 Tallapoosa River & USGS 02413300 Little Tallapoosa River for reservoir inflow; USGS 02415000 Hillabee Creek

^b Incomplete data used to calculate; data unavailable

Table 2. Site level CPE's for crayfish species collected in regulated and unregulated reaches of the Tallapoosa River basin by year. Annual values include *Procambarus spiculifer*, *Cambarus englishi*, *C. halli*, but exclude YOY in calculations.

Site	2005	2006	2007	2008	2009	Overall Rank	Overall Rank (excluding YOY)
Dam to Malone A	0.10	0.13	0.20	0.45	0.25	18	20
Dam to Malone B	0.45	0.30	0.50	1.75	1.95	4	2
Dam to Malone C	0.30	0.45	0.75	0.55	1.35	7	4
Dam to Malone D	0.18	0.40	0.18	0.00	0.75	16	13
Dam to Malone E	0.68	0.38	0.55	0.25	0.85	8	5
Malone to Wadley A	0.23	0.35	0.03	0.35	0.55	15	14
Malone to Wadley B	0.08	0.13	0.20	0.60	0.60	14	15
Malone to Wadley C	0.18	1.00	0.70	0.40	1.10	6	3
Malone to Wadley D	0.20	0.53	0.20	0.00	0.55	13	11
Malone to Wadley E	0.20	0.15	0.25	0.05	0.50	20	18
Horseshoe Bend A	0.15	0.30	0.15	0.10	0.20	21	21
Horseshoe Bend B	0.10	0.50	0.00	0.00	0.10	22	22
Horseshoe Bend C	0.40	0.35	0.00	0.15	0.20	19	17
Horseshoe Bend D	0.40	0.00	0.00	0.00	0.00	23	23
Horseshoe Bend E	0.25	0.05	0.05	0.00	0.00	24	24
Hillabee Creek A	0.10	0.35	0.29	0.05	0.20	17	19
Hillabee Creek B	0.00	0.03	0.06	0.00	0.00	25	25
Hillabee Creek C	0.90	0.33	.	0.00	0.10	9	8
Hillabee Creek D	0.50	0.55	.	0.10	0.00	10	7
Hillabee Creek E	0.50	0.50	0.06	0.05	0.00	12	9
Upper Tallapoosa A	0.25	0.23	0.55	.	0.25	3	10
Upper Tallapoosa B	0.15	0.38	0.30	.	0.25	2	12
Upper Tallapoosa C	0.45	0.90	0.70	.	0.95	1	1
Upper Tallapoosa D	0.25	0.30	0.53	1.05	0.25	5	6
Upper Tallapoosa E	0.05	0.18	0.24	0.65	0.15	11	16

Table 3.- Catch data summary for crayfishes by species and by year from sites in the Tallapoosa River basin. Sites are arranged in a longitudinal fashion descending from Harris Dam. Sites below the solid black line are unregulated sites (Hillabee Creek between Sanford Road and Alabama Hwy 22 and the Upper Tallapoosa River between Ben Mills and Evans Ferry). Individuals <14 mm were unidentified and classified as YOY. Dashes indicate no collections made due to site inaccessibility.

Site	Species			YOY	Adult*	Year					Total
	<i>P. spiculifer</i>	<i>C. englishi</i>	<i>C. halli</i>			2005	2006	2007	2008	2009	
Dam to Malone A	24	3	1	12	3	12	5	8	13	5	43
Dam to Malone B	65	38	7	46	14	27	13	30	61	39	170
Dam to Malone C	53	32	7	8	5	19	17	31	11	27	105
Dam to Malone D	25	14	6	2	0	9	16	7	0	15	47
Dam to Malone E	39	38	6	5	3	29	15	23	7	17	91
Malone to Wadley A	26	13	0	9	3	12	15	2	10	12	51
Malone to Wadley B	14	20	5	15	1	3	5	16	18	13	55
Malone to Wadley C	25	60	19	39	1	10	41	62	9	22	144
Malone to Wadley D	18	25	5	8	0	9	21	15	0	11	56
Malone to Wadley E	27	4	4	2	0	10	6	10	1	10	37
Horseshoe Bend A	5	4	7	4	0	3	6	4	3	4	20
Horseshoe Bend B	6	1	6	1	0	2	10	0	1	1	14
Horseshoe Bend C	9	8	3	2	0	8	7	0	4	3	22
Horseshoe Bend D	6	2	0	2	0	9	0	0	0	1	10
Horseshoe Bend E	5	2	0	2	0	5	1	1	1	1	9
Hillabee Creek A	21	1	2	6	0	3	16	6	1	4	30
Hillabee Creek B	2	0	0	3	0	1	1	1	1	1	5
Hillabee Creek C	28	3	2	18	0	23	24	-	0	4	51
Hillabee Creek D	29	2	2	11	1	10	32	-	2	1	45
Hillabee Creek E	26	0	5	7	1	14	21	1	1	2	39
Upper Tallapoosa A	19	3	8	86	0	7	77	15	-	17	116
Upper Tallapoosa B	21	1	7	112	0	3	86	9	-	43	141
Upper Tallapoosa C	41	10	27	97	0	9	105	21	-	40	175
Upper Tallapoosa D	19	13	17	59	10	5	60	19	25	9	118
Upper Tallapoosa E	19	2	5	26	4	2	29	7	15	3	56
Total	572	299	151	582	46	244	629	288	184	305	1650

*Note: refers to lack of species specific data for summer 2008.

Table 4.- Covariates used to model detection and occupancy of crayfish species collected from regulated and unregulated reaches of the Tallapoosa River basin.

Detection	Occupancy
Depth	Regulated
Velocity	Distance
Vegetation	Reach ^a
Substrate	

^aA posteriori model

Table 5.- Top detection models for crayfish species collected with PAE's from regulated and unregulated sites of the Tallapoosa River basin.

Species	Year	Top Detection Model
<i>Procambarus spiculifer</i>	2005	psi(.),p(depth + vegetation)
	2006	psi(.),p(depth + vegetation)
	2007	psi(.),p(depth + vegetation)
	2008	psi(.),p(velocity + substrate)
	2009	psi(.),p(velocity + vegetation)
<i>Cambarus englishi</i>	2005	psi(.),p(depth + vegetation + substrate)
	2006	psi(.),p(vegetation + substrate)
	2007	psi(.),p(velocity + vegetation + substrate)
	2008	psi(.),p(vegetation + substrate)
	2009	psi(.),p(vegetation + substrate)
<i>Cambarus halli</i>	2005	psi(.),p(depth + vegetation)
	2006	psi(.),p(velocity + depth + vegetation)
	2007	psi(.),p(velocity + depth + vegetation)
	2008	psi(.),p(depth + vegetation + substrate)
	2009	psi(.),p(depth + vegetation + substrate)
YOY	2005	psi(.),p(depth + vegetation)
	2006	psi(.),p(velocity + vegetation + substrate)
	2007	psi(.),p(velocity + vegetation + substrate)
	2008	psi(.),p(vegetation)
	2009	psi(.),p(vegetation + substrate)

Table 6.- Occupancy model set for *P. spiculifer* collected from regulated and unregulated sites in the Tallapoosa River basin with K-L confidence model set final weights.

Model	AIC	Δ AIC	AIC weight	K	-2log-likelihood	Final weight
2005						
psi(.),p(depth + vegetation)	537.07	0.00	0.57	4	529.07	1.00
psi(Regulated),p(depth + vegetation)	539.07	2.00	0.21	5	529.07	-
psi(Distance),p(depth + vegetation)	539.07	2.00	0.21	5	529.07	-
psi(Reach),p(depth + vegetation)	545.07	8.00	0.01	8	529.07	-
psi(.),p(.)	548.73	11.66	0.00	2	544.73	-
2006						
psi(.),p(depth + vegetation)	764.67	0.00	0.50	4	756.67	0.90
psi(Regulated),p(depth + vegetation)	766.06	1.39	0.25	5	756.06	-
psi(Distance),p(depth + vegetation)	766.67	2.00	0.18	5	756.67	-
psi(Reach),p(depth + vegetation)	768.44	3.77	0.08	8	752.44	0.10
psi(.),p(.)	799.53	34.86	0.00	2	795.53	-
2007						
psi(Reach),p(depth + vegetation)	468.36	0.00	0.86	8	452.36	0.92
psi(.),p(depth + vegetation)	473.28	4.92	0.07	4	465.28	0.08
psi(Regulated),p(depth + vegetation)	474.65	6.29	0.04	5	464.65	-
psi(Distance),p(depth + vegetation)	475.04	6.68	0.03	5	465.04	-
psi(.),p(.)	495.33	26.97	0.00	2	491.33	-
2008						
psi(Reach),p(velocity + substrate)	129.24	0.00	0.98	8	113.24	1.00
psi(Distance),p(velocity + substrate)	138.18	8.94	0.01	5	128.18	-
psi(.),p(velocity + substrate)	139.31	10.07	0.01	4	131.31	-
psi(Regulated),p(velocity + substrate)	140.66	11.42	0.00	5	130.66	-
psi(.),p(.)	144.90	15.66	0.00	2	140.90	-
2009^a						
psi(Reach),p(velocity + vegetation)	316.06	0.00	0.86	8	357.07	0.88
psi(.),p(velocity + vegetation)	321.18	5.12	0.07	4	372.69	0.07
psi(Distance),p(velocity + vegetation)	321.71	5.65	0.05	5	370.93	0.05
psi(Regulated),p(velocity + vegetation)	322.97	6.91	0.03	5	372.44	-
psi(.),p(.)	339.09	23.03	0.00	2	398.76	-

^a AIC values replaced by QAIC using variance inflation factor c-hat to adjust for overdispersion

Table 7.- Occupancy model sets for *C. englishi* collected from regulated and unregulated sites in the Tallapoosa River basin with K-L confidence model set final weights.

Model	AIC	Δ AIC	AIC weight	K	-2log-likelihood	Final weight
2005^a						
psi(.),p(depth + vegetation + substrate)	251.92	0.00	0.50	5	290.31	1.00
psi(Distance),p(depth + vegetation + substrate)	253.17	1.25	0.27	6	289.41	-
psi(Regulated),p(depth + vegetation + substrate)	253.73	1.81	0.20	6	290.07	-
psi(Reach),p(depth + vegetation + substrate)	257.56	5.64	0.03	9	287.47	-
psi(.),p(.)	268.25	16.33	0.00	2	317.09	-
2006						
psi(Regulated),p(vegetation + substrate)	351.83	0.00	0.43	5	341.83	0.75
psi(Distance),p(vegetation + substrate)	351.83	0.00	0.19	5	341.83	-
psi(.),p(vegetation + substrate)	355.22	3.39	0.08	4	347.22	0.14
psi(Reach),p(vegetation + substrate)	355.70	3.87	0.06	8	339.70	0.11
psi(.),p(.)	403.12	51.29	0.00	2	399.12	-
2007						
psi(Distance),p(velocity + vegetation + substrate)	275.49	0.00	0.29	6	263.49	0.29
psi(Reach),p(velocity + vegetation + substrate)	275.60	0.11	0.27	9	257.60	0.27
psi(.),p(velocity + vegetation + substrate)	275.79	0.30	0.25	5	265.79	0.25
psi(Regulated),p(velocity + vegetation + substrate)	276.35	0.86	0.19	6	264.35	0.19
psi(.),p(.)	299.37	23.88	0.00	2	295.37	-
2008						
psi(.),p(vegetation + substrate)	140.90	0.00	0.49	4	132.90	0.85
psi(Regulated),p(vegetation + substrate)	142.26	1.36	0.25	5	132.26	-
psi(Distance),p(vegetation + substrate)	142.88	1.98	0.18	5	132.88	-
psi(Reach),p(vegetation + substrate)	144.42	3.52	0.08	8	128.42	0.15
psi(.),p(.)	155.57	14.67	0.00	2	151.57	-
2009^a						
psi(Distance),p(vegetation + substrate)	234.68	0.00	0.48	5	292.08	0.48
psi(Regulated),p(vegetation + substrate)	236.08	1.40	0.24	5	293.90	0.24
psi(Reach),p(vegetation + substrate)	236.88	2.20	0.16	8	287.15	0.16
psi(.),p(vegetation + substrate)	237.38	2.70	0.12	4	298.20	0.12
psi(.),p(.)	246.88	12.20	0.00	2	315.75	-

^a AIC values replaced by QAIC using variance inflation factor c-hat to adjust for overdispersion

Table 8.- Occupancy model sets for *C. halli* collected from regulated and unregulated sites in the Tallapoosa River basin with K-L confidence model set final weights.

Model	AIC	Δ AIC	AIC weight	K	-2log-likelihood	Final weight
2005						
psi(.),p(depth + vegetation)	174.62	0.00	0.53	4	166.62	1.00
psi(Regulated),p(depth + vegetation)	176.62	2.00	0.19	5	166.62	-
psi(Distance),p(depth + vegetation)	176.62	2.00	0.19	5	166.62	-
psi(.),p(.)	178.53	3.91	0.07	2	174.53	-
psi(Reach),p(depth + vegetation)	182.62	8.00	0.01	8	166.62	-
2006						
psi(.),p(velocity + depth + vegetation)	356.08	0.00	0.51	5	346.08	0.90
psi(Regulated),p(velocity + depth + vegetation)	357.67	1.59	0.23	6	345.67	-
psi(Distance),p(velocity + depth + vegetation)	357.94	1.86	0.20	6	345.94	-
psi(Reach),p(velocity + depth + vegetation)	360.54	4.46	0.05	9	342.54	0.10
psi(.),p(.)	366.98	10.90	0.00	2	362.98	-
2007						
psi(Reach),p(velocity + depth + vegetation + substrate)	248.19	0.00	0.95	10	228.19	1.00
psi(.),p(velocity + depth + vegetation + substrate)	255.39	7.20	0.03	6	243.39	-
psi(Regulated),p(velocity + depth + vegetation + substrate)	256.25	8.06	0.02	7	242.25	-
psi(Distance),p(velocity + depth + vegetation + substrate)	257.39	9.20	0.01	7	243.39	-
psi(.),p(.)	280.26	32.07	0.00	2	276.26	-
2008						
psi(Reach),p(depth + vegetation + substrate)	61.52	0.00	0.47	9	43.52	0.48
psi(Distance),p(depth + vegetation + substrate)	63.00	1.48	0.22	6	51.00	0.22
psi(.),p(depth + vegetation + substrate)	63.45	1.93	0.18	5	53.45	0.18
psi(Regulated),p(depth + vegetation + substrate)	64.18	2.66	0.12	6	52.18	0.12
psi(.),p(.)	68.95	7.43	0.01	2	64.95	-
2009						
psi(Reach),p(depth + vegetation + substrate)	138.80	0.00	0.47	9	120.80	0.49
psi(.),p(depth + vegetation + substrate)	140.52	1.72	0.20	5	130.52	0.20
psi(Distance),p(depth + vegetation + substrate)	140.64	1.84	0.19	6	128.64	0.19
psi(Regulated),p(depth + vegetation + substrate)	141.51	2.71	0.12	6	129.51	0.12
psi(.),p(.)	145.77	6.97	0.01	2	141.77	-

Table 9.- Reach and basin occupancy estimates (Ψ) and naïve occupancy for *P. spiculifer* collected from regulated and unregulated reaches of the Tallapoosa River basin.

Reaches	2005			2006			2007			2008			2009		
	Naïve	Ψ	SE	Naïve	Ψ	SE	Naïve	Ψ	SE	Naïve	Ψ	SE	Naïve	Ψ	SE
All Sites	0.96	1	a	0.92	0.99	0.06	0.78	0.85	0.09	0.41	0.42	0.10	0.72	0.78	0.09
Dam to Malone	-	-		1.00	0.23		0.99	0.04		0.62	0.51		0.98	0.05	
Malone to Wadley	-	-		1.00	0.23		0.99	0.04		0.82	0.27		0.98	0.05	
Horseshoe Bend	-	-		0.96	0.31		0.27	0.46		0	0		0.29	0.47	
Hillabee Creek	-	-		1.00	0.23		0.72	0.52		0	0		0.45	0.48	
Upper Tallapoosa	-	-		1.00	0.23		0.99	0.04		1.00	0		0.97	0.07	

^a Variance -covariance not computed successfully

Table 10.- Reach and basin occupancy estimates (Ψ) and naïve occupancy for *C. englishi* collected from regulated and unregulated reaches of the Tallapoosa River basin.

Reaches	2005			2006			2007			2008			2009		
	Naïve	Ψ	SE	Naïve	Ψ	SE	Naïve	Ψ	SE	Naïve	Ψ	SE	Naïve	Ψ	SE
All Sites	0.56	0.68	0.12	0.64	0.83	0.12	0.52	0.68	0.13	0.5	0.58	0.13	0.64	0.78	0.11
Dam to Malone	-	-		0.97	0.08		0.84	0.30		0.60	0.41		0.98	0.26	
Malone to Wadley	-	-		0.97	0.08		0.71	0.48		0.59	0.41		0.97	0.27	
Horseshoe Bend	-	-		0.96	0.10		0.60	0.50		0.53	0.41		0.84	0.43	
Hillabee Creek	-	-		0.52	0.54		0.33	0.43		0.58	0.40		0.47	0.47	
Upper Tallapoosa	-	-		0.59	0.54		0.58	0.50		0.65	0.36		0.58	0.46	

Table 11.- Reach and basin occupancy estimates (Ψ) and naïve occupancy for *C. halli* collected from regulated and unregulated reaches of the Tallapoosa River basin.

Reaches	2005			2006			2007			2008			2009		
	Naïve	Ψ	SE	Naïve	Ψ	SE	Naïve	Ψ	SE	Naïve	Ψ	SE	Naïve	Ψ	SE
All Sites	0.60	1	A	0.68	0.85	0.12	0.61	0.71	0.12	0.23	0.61	0.29	0.40	0.58	0.15
Dam to Malone	-	-	-	0.87	0.33	-	1.00	0	-	0.70	0.42	-	0.77	0.46	-
Malone to Wadley	-	-	-	0.84	0.39	-	0.87	0.15	-	0.24	0.43	-	0.66	0.57	-
Horseshoe Bend	-	-	-	0.83	0.39	-	0	0	-	0.79	0.42	-	0.32	0.42	-
Hillabee Creek	-	-	-	0.84	0.37	-	0.43	0.23	-	0.45	0.41	-	0.24	0.37	-
Upper Tallapoosa	-	-	-	0.87	0.33	-	1.00	0	-	0.93	0.36	-	0.54	0.59	-

^A Variance -covariance not computed successfully



Figure 1.- Map of sampling sites located in the Tallapoosa River basin including Randolph, Cleburne, and Tallapoosa counties, Alabama. Headwaters are located in Georgia.

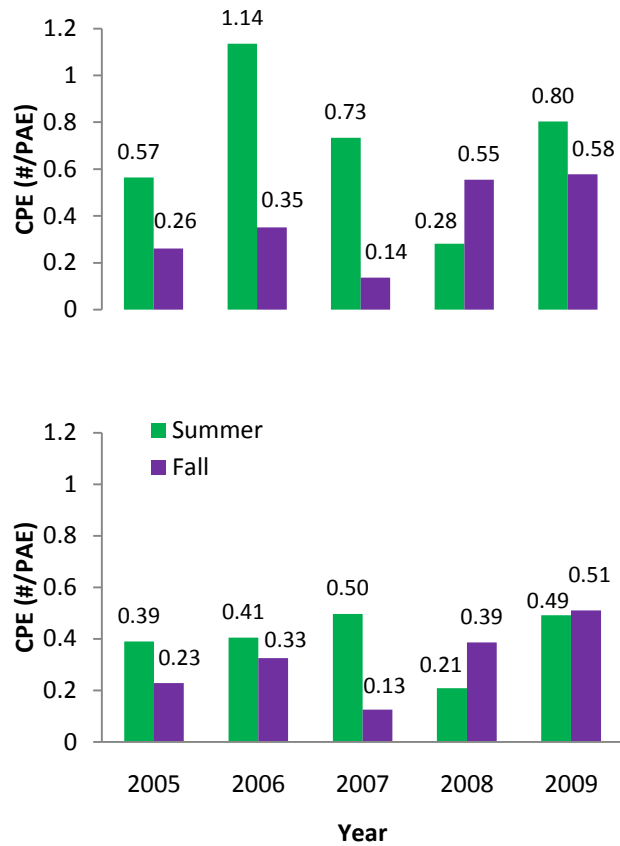


Figure 2.- Mean catch-per-effort (# of individuals/pre-positioned area electrofisher) of crayfishes captured in the Tallapoosa River basin pooled across sites. Top graphs include *P. spiculifer*, *C. englishi*, *C. halli*, and unidentified YOY; bottom graphs exclude unidentified YOY.

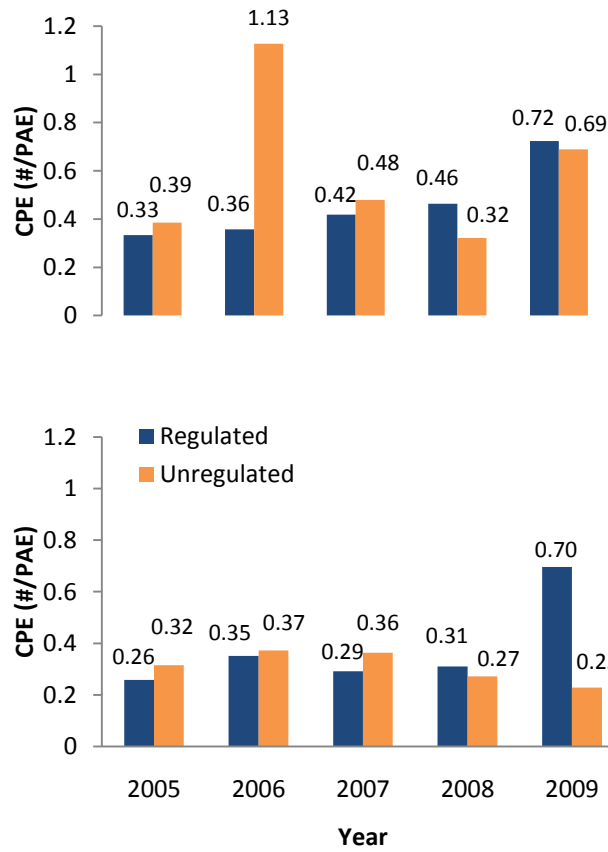


Figure 3.- Mean catch-per-effort (# of individuals/pre-positioned area electrofisher) of crayfishes captured in the Tallapoosa River basin pooled across sites by regulation group. Top graphs include *P. spiculifer*, *C. englishi*, *C. halli*, and unidentified YOY; bottom graphs exclude unidentified YOY.

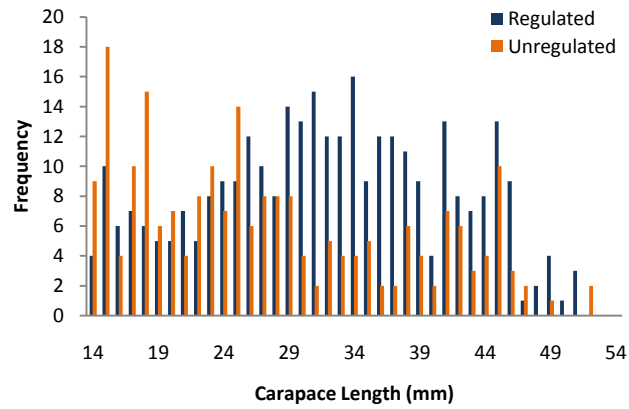


Figure 4.- Carapace length frequency histogram for *P. spiculifer* captured between 2005-2009 in the Tallapoosa River basin by regulation group.

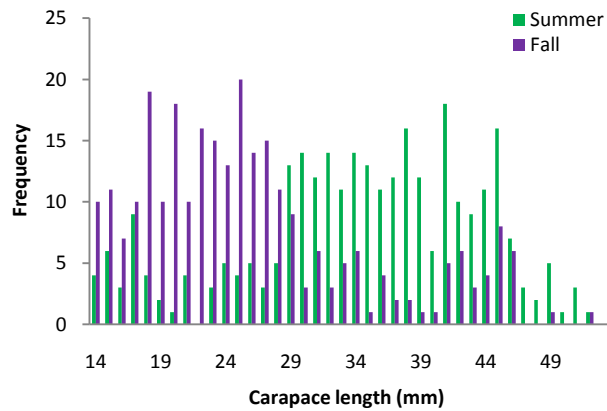


Figure 5.- Carapace length frequency histogram for *P. spiculifer* captured between 2005-2009 in regulated and unregulated reaches of the Tallapoosa River basin by season.

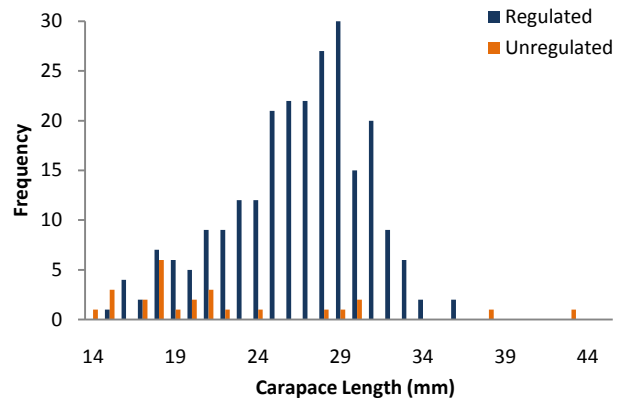


Figure 6.- Carapace length frequency histogram for *C. englishi* captured between 2005-2009 in the Tallapoosa River basin by regulation group.

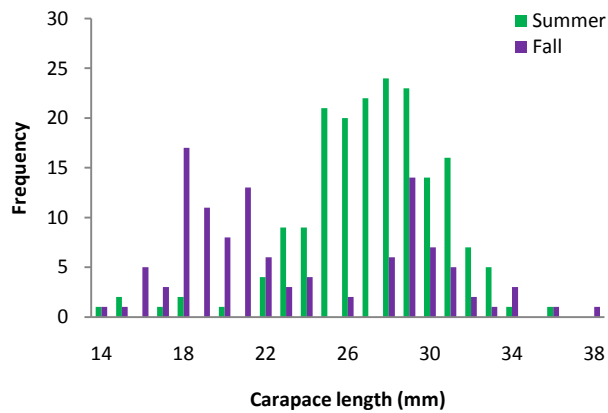


Figure 7.- Carapace length frequency histogram for *C. englishi* captured between 2005-2009 in regulated and unregulated reaches of the Tallapoosa River basin by season.

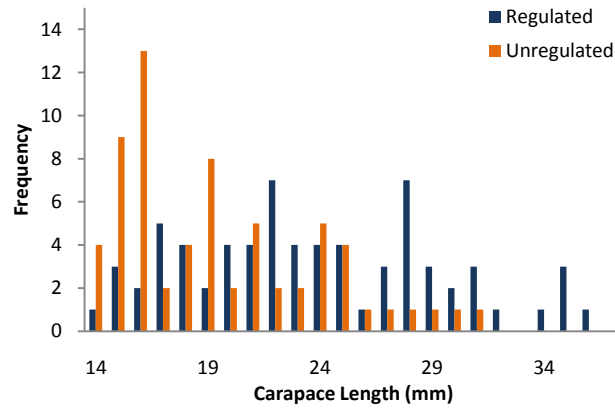


Figure 8.- Carapace length frequency histogram for *C. halli* captured between 2005-2009 in the Tallapoosa River basin by regulation group.

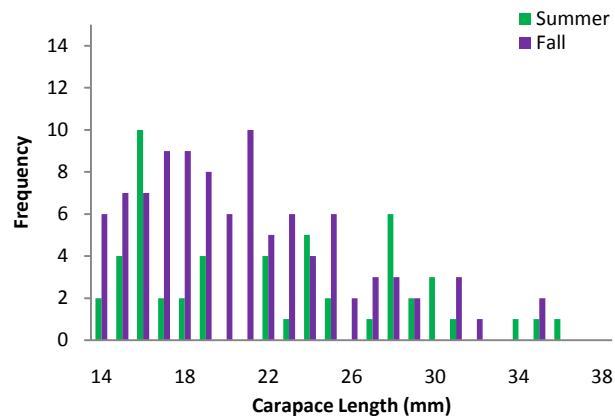


Figure 9.- Carapace length frequency histogram for *C. halli* captured between 2005-2009 in regulated and unregulated reaches of the Tallapoosa River basin by season.

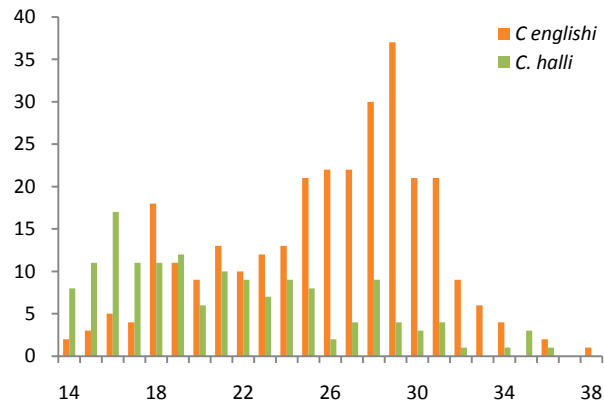


Figure 10.- Carapace length frequency histogram of 2 species of crayfish collected between 2005-2009 in regulated and unregulated reaches of the Tallapoosa River basin.

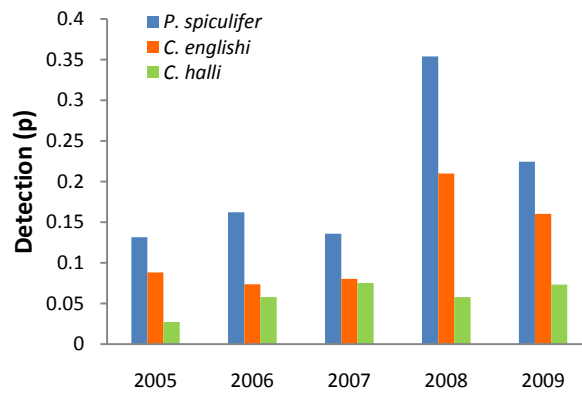


Figure 11.- Average detection probability values calculated from top detection model for 3 species of crayfishes collected from regulated and unregulated sites in the Tallapoosa River basin.

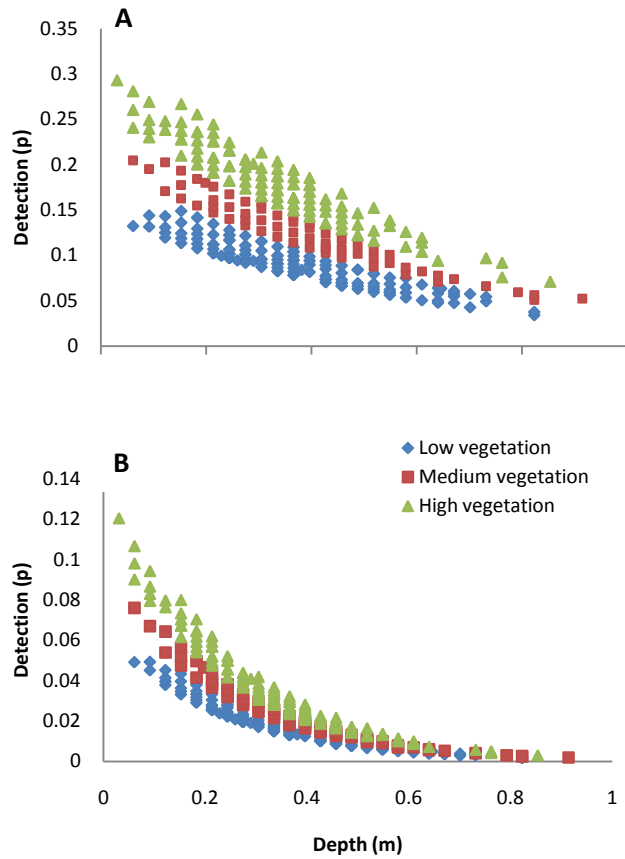


Figure 12.- Relation between detection and depth for 3 levels of vegetation (low = 0-30%, medium = 31-60%, and high = 61-100% areal coverage) for 2 species of crayfishes, (A) *P. spiculifer* and (B) *C. halli*, collected from regulated and unregulated sites in the Tallapoosa River Basin. Data are from 2005 top detection models.

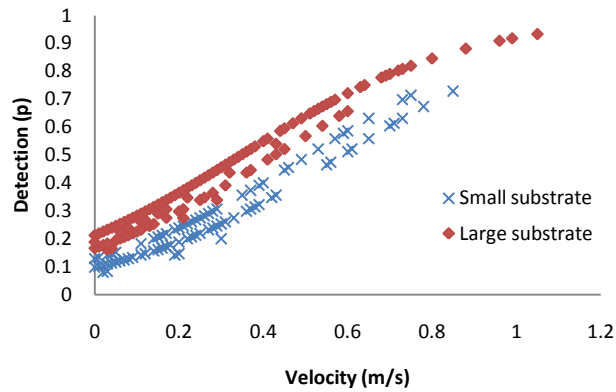


Figure 13.- Relation between detection and velocity for 2 classes of substrate for *P. spiculifer* collected from regulated and unregulated sites in the Tallapoosa River basin. Data are from 2008 top detection model with small substrate having values ≤ 3 and large indicating values > 3 (see Methods for substrate values).

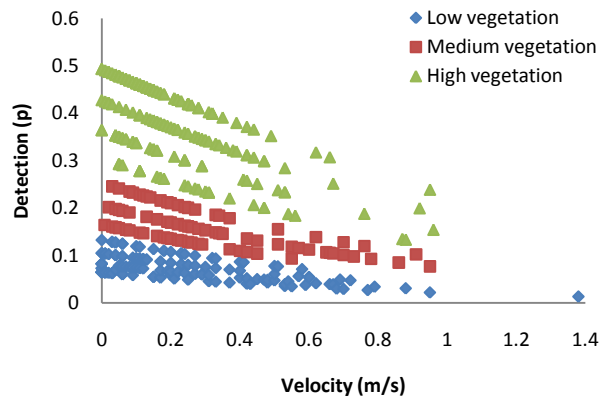


Figure 14.- Relation between detection and velocity for 3 levels of vegetation (low = 0-30%, medium = 31-60%, and high = 61-100% areal coverage) for *P. spiculifer* collected from regulated and unregulated sites in the Tallapoosa River Basin. Data are from 2009 top detection model.

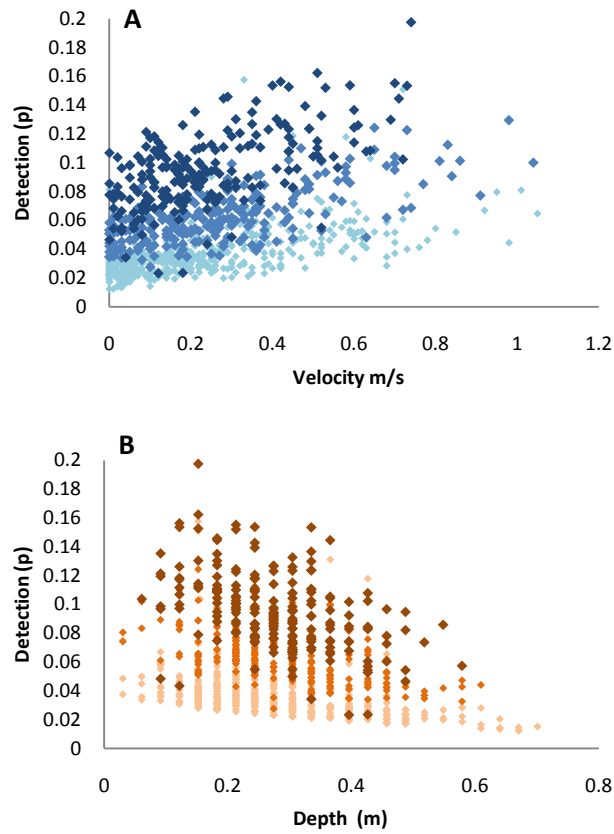


Figure 15.- Relation between detection and (A) velocity and depth (B) for *C. halli* collected from regulated and unregulated sites in the Tallapoosa River Basin. Data are from 2006 top detection model where marker color indicates vegetation level (low = 0-30%, medium = 31-60%, and high = 61-100% areal coverage); the darkest markers indicating high vegetation.

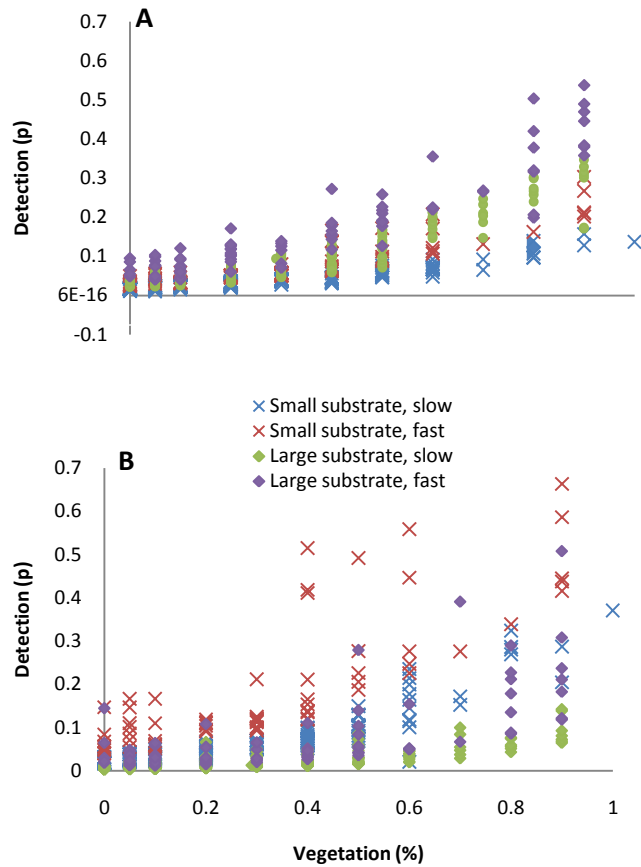


Figure 16.- Relation between detection and percent vegetation with the influence of substrate size and slow (<20cm/s) and fast (>21 cm/s) velocities for crayfish collected from regulated and unregulated sites in the Tallapoosa River basin. Small substrate includes sand and gravel; large substrate including small and large woody debris, cobbles, boulders, and bedrock shelf. Data are from 2007 detection models; (A) *C. englishi* and (B) YOY.

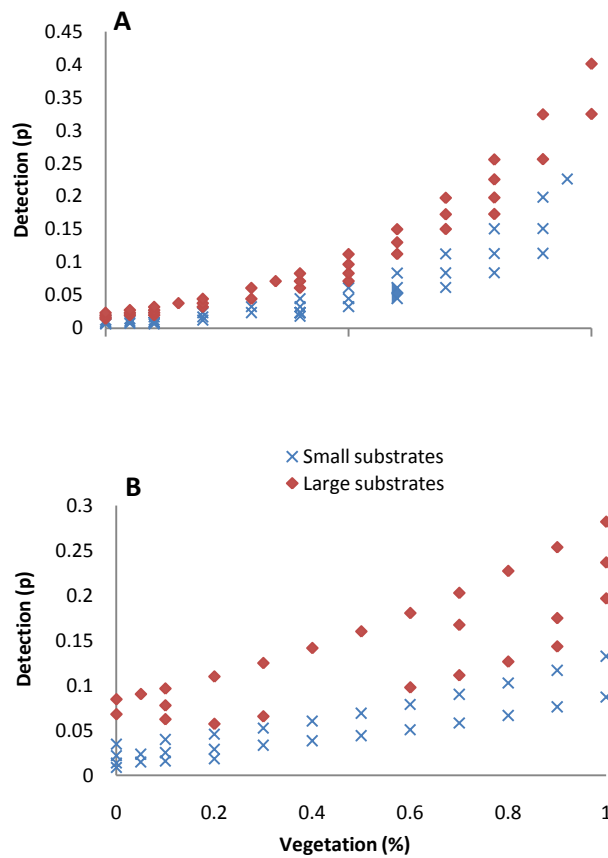


Figure 17.- Relation between detection and vegetation for 2 classes of substrate (small substrates having a value ≤ 3 and large indicating values > 3 ; see Methods for substrate values) for *C. englishi* collected from regulated and unregulated sites in the Tallapoosa River Basin. Data are from top detection models of (A) 2006 and (B) 2009. Notice in 2006 detection was more variable and vegetation had a stronger influence whereas in 2009 substrate size had a greater influence.

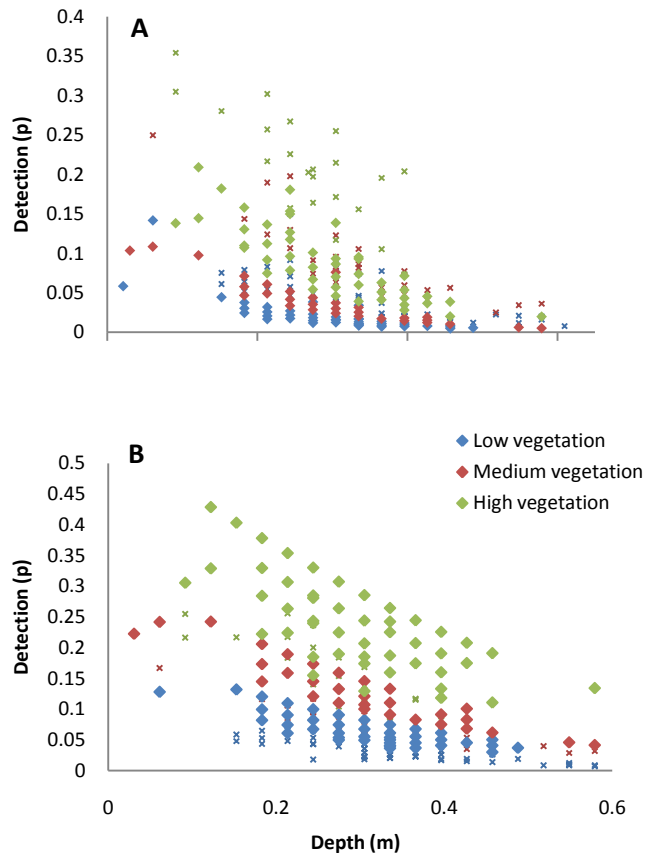


Figure 18.- Relation between detection and depth for 3 levels of vegetation (low = 0-30%, medium = 31-60%, and high = 61-100% areal coverage) and 2 classes of substrates for (A) *C. halli* and (B) *C. englishi*. Smaller markers plot represent detection values over substrates with a value ≤ 3 and larger markers indicate detection values > 3 (see Methods for substrate values). Data for *C. englishi* and *C. halli* are from 2005 and 2009 top detection models respectively, collected from regulated and unregulated sites in the Tallapoosa River Basin.

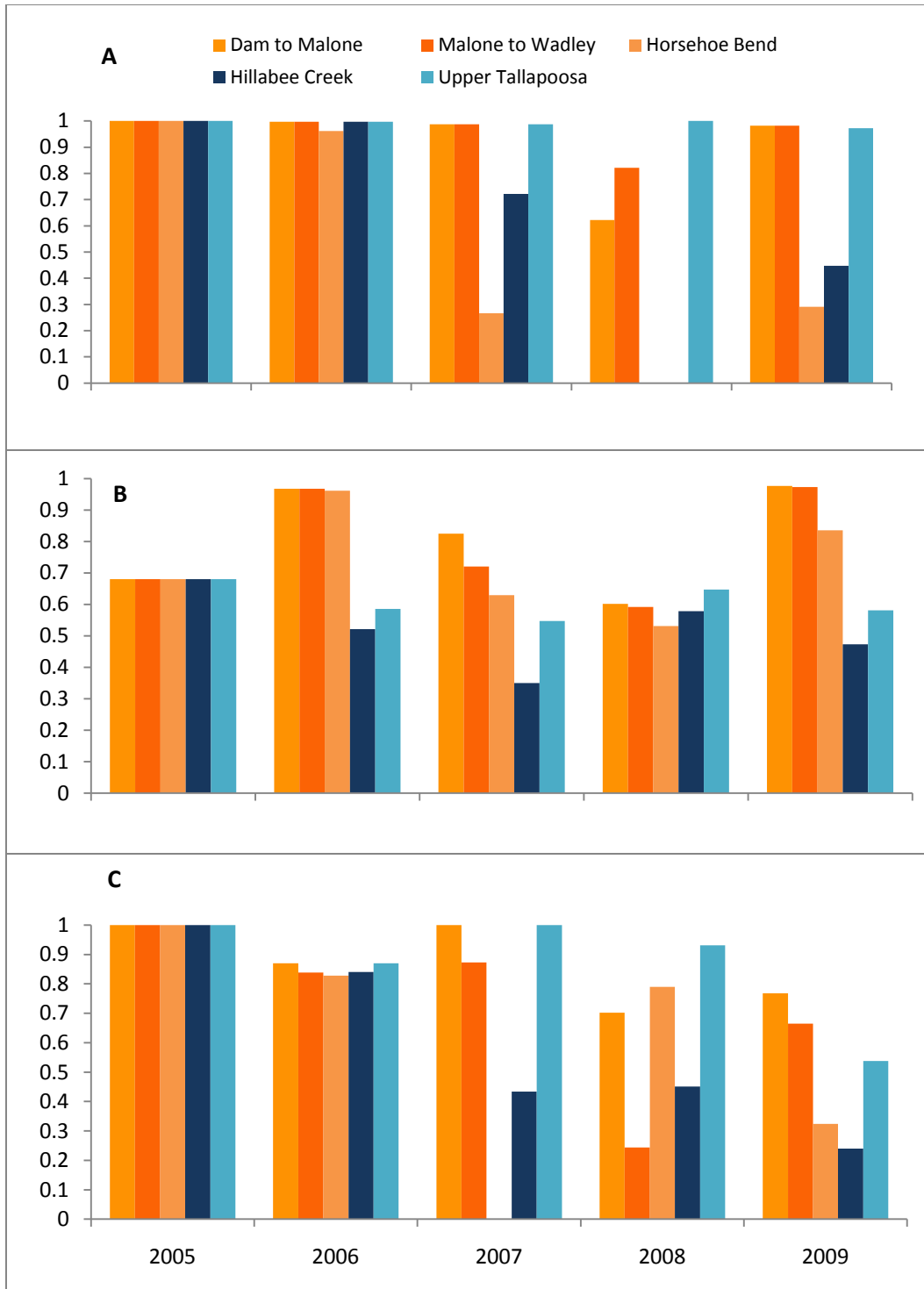


Figure 19.- Model averaged estimates of reach scale occupancy by year and species; (A) *P. spiculifer*, (B) *C. englishi* and (C) *C. halli*.

REFERENCES

- Bailey, L. L., T.R. Simmons, and K. H. Pollock. 2004. Estimating site occupancy and species detection probability parameters for terrestrial salamanders. *Ecological Applications* 14(3):692-702.
- Bain M.B., J.T. Finn, and H.E. Booke. 1988. Stream flow regulation and fish community structure. *Ecology* 69(2):382-392.
- Bouchard, R. W. 1978. Taxonomy, distribution, and general ecology of the genera of North American crayfishes. *Fisheries* 3(6):11-19.
- Boulinier, T., J.D. Nichols, J. D. Sauer, J. E. Hines, and K.H. Pollock 1998. Estimating species richness: the importance of heterogeneity in species detectability. *Ecology* 79 (3): 1018-1028.
- Brewer, S. K., R.J. DiStefano, and C.F. Rabeni, 2009. The influence of age-specific habitat selection by a stream, crayfish community (*Orconectes* spp.) on secondary production. *Hydrobiologia* 619: 1-10.
- Bubb, D.H., T.J. Thom, and M.C. Lucas. 2004. Movement and dispersal of the invasive signal crayfish *Pacifistica leniusculus* in upland rivers. *Freshwater Biology* 49: 357-368.
- Burnham, K. P., and D. R. Anderson. 2002. Model selection and inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, New York, USA.
- Comprehensive Wildlife Conservation Strategy (Draft). 2005. Alabama Department of Conservation and Natural Resources. <http://www.outdooralabama.com/research-mgmt/cwcs/>.
- Dennard, S., J.T. Peterson, and E.S. Hawthorne. 2009. Life History and Ecology of *Cambarus halli*. *Southeastern Naturalist* 8 (3):479-494.
- DiStefano, R.J, J.J. Decoske, T.M. Vangilder and L.S. Barnes. 2003. Macrohabitat partitioning among three crayfish species in two Missouri streams U.S.A. *Crustaceana* 76 (3): 343-362.

- DiStefano, R.J., C.M. Gale, B.A. Wagner, and R.D. Zweifel. 2003b. A sampling method to assess lotic crayfish communities. *Journal of Crustacean Biology* 23(3): 678-690.
- Englund, G. and J. J. Krupa. 2000. Habitat use by crayfish in stream pools: influence of predators, depth and body size. *Freshwater Biology* 43:75-83
- Flinders C.A., and D.D Magoulick. 2007. Habitat use and selection within Ozark lotic crayfish assemblages: spatial and temporal variation. *Journal of Crustacean Biology* 27 (7):242-254.
- Folkerts, G. W. 1997. State and fate of the world's aquatic fauna. Pages 1–16 in G. W. Benz and D. E. Collins, editors. *Aquatic fauna in peril: the southeastern perspective*. Special publication 1. Southeast Aquatic Research Institute, Lenz Design and Communications, Decatur, Georgia.
- France, R., J. Holmes, and A. Lynch. 1991. Use of size-frequency data to estimate the age composition of crayfish populations. *Canadian Journal of Fisheries and Aquatic Sciences* 48:2324–2332.
- Freeman, M.C., Z.H. Bowen, K.D. Bovee, and E. R. Irwin. 2001. Flow and Habitat effects of YOY fish abundance in natural and altered flow regimes. *Ecological Applications* 11 (1) : 179-190.
- Fortino, K. and R. P. Creed. 2007. Abiotic factors, competition or predation what determines distribution of crayfish in a watershed *Hydrobiologia* 575:301-314
- Gore, J.A. and R.M. Bryant. 1990. Temporal shifts in physical habitat of the crayfish, *Orconectes neglectus* (Faxon). *Hydrobiologia* 199: 131-142.
- Gu, W. and R. K. Swihart. 2004. Absent or undetected? Effects of non-detection of species occurrence on wildlife-habitat models. *Biological Conservation* 116: 195-203.
- Hayer, C.A., Irwin, E.R. Influence of Gravel Mining and Other Factors on Detection Probabilities of Coastal Plain Fishes in the Mobile River Basin, Alabama *Transactions of the American Fisheries Society* 137(6):1606-1620
- Helms, B.S and R.P. Creed 2005. The effects of two coexisting crayfish on an Appalachian river community. *Journal of the North American Benthological Society*. 24 (1):113-122.
- Hobbs, Jr., H. H. 1942. *The crayfishes of Florida*. University of Florida Publications, Biological Science Series No. 3: 1-79.
- Hobbs, Jr., H. H. 1981. *The crayfishes of Georgia*. Smithsonian Institution, Washington.

- Huryn, A.D. and J.B. Wallace. 1987. Production and litter processing by crayfish in an Appalachian mountain stream. *Freshwater Biology* 18: 227-286.
- Hyne, R.V., and W.A. Maher. 2003. Invertebrate biomarkers: Links to toxicosis that predict population decline. *Ecotoxicology and Environmental Safety* 54:366–374.
- Irwin, E.R., and M.C. Freeman. 2002. Proposal for adaptive management to conserve biotic integrity in a regulated segment of the Tallapoosa River, Alabama, U.S.A. *Conservation Biology*. 16(5):1212-1222.
- Irwin, E.R., Kennedy, K.M., Goar, T.P., Martin, B.M. Martin, and M.M. Martin. 2009 Adaptive management and monitoring for restoration and faunal recolonization of Tallapoosa River shoal habitats: Interim report. <http://www.oudooralabama.com/research-mgmt/State%20Wildlife%20Grants/projectsfunded.cfm>
- Jones, S.N. and E.A. Bergey. 2007. Habitat segregation in stream crayfishes: implication for conservation. *Journal of the North American Benthological Society* 26 (1):134-144.
- Kershner, M. W. and D. M. Lodge 1995 Effects of littoral habitat and fish predation on the distribution of an exotic crayfish, *Orconectes rusticus*. *Journal of the North American Benthological Society* 14(3): 414-422
- Kendall, W.L. and G.C. White. 2009 A cautionary note on substituting spatial subunits for repeated temporal sampling studies of site occupancy. *Journal of Applied Ecology* 46:1182-1188.
- Kinsolving A.D. and M.B. Bain. 1993. Fish assemblage recovery along a riverine disturbance gradient. *Ecological Applications* 3(3):531-544
- MacKenzie, D. I., J.D. Nichols, G.B. Lachman, S. Droege, J.A. Royle, and C.A. Langtimm 2002. Estimating site occupancy rates when detection probabilities are less than one. *Ecology* 83 (8): 2248-2255.
- MacKenzie, D. I., and L. Bailey. 2004. Assessing the Fit of Site Occupancy Models. *Journal of Agricultural, Biological and Environmental Statistics* 9:300–318.
- MacKenzie, D. I., J.D. Nichols, J.A. Royle, K.H. Pollock, L.L. Bailey, and J.E. Hines 2006. *Occupancy estimation and modeling*. Elsevier, Burlington, MA.
- Maude, S.H. and D.D. Williams. 1983. Behavior of crayfish in water currents: hydrodynamics of eight species with reference to their distribution patterns in southern Ontario. *Canadian Journal of Fisheries and Aquatic Science* 40: 68-77.
- Minckley, W.L. and J.E. Craddock. 1961. Active predation of crayfish on fishes. *The Progressive Fish- Culturist* 23: 120-123.

- Mitchell D.J. and L.A. Smock. 1991. Distribution, life history and production of crayfish in the James River, Virginia. *American Midland Naturalist* 126:353-363.
- Momot W. T. 1995. Redefining the role of crayfish in aquatic ecosystems. *Review in Fisheries Science* 3(1): 33-63.
- Nichols, J. D., Boulinier, T, J. E. Hines, K.H. Pollock, and J. D. Sauer 1998. Estimating rates of local species extinction, colonization, and turnover in animal communities. *Ecological Applications* 8 (4): 1213-1255.
- Nichols, J. D., Boulinier, T, J. E. Hines, K.H. Pollock, and J. D. Sauer 1998. Inference methods for spatial variation in species richness and community composition when not all species are detected
- Ollsson K, and P. Nystrom. 2009. Non-interactive effects of habitat complexity and adult crayfish on survival and growth of YOY crayfish (*Pacifastacus leniusculus*). *Freshwater Biology* 54: 35-46.
- Peters, R.H. 1983. *The ecological implications of body size*. Cambridge University Press, Cambridge, England.
- Peterson, J.T. and J. Dunham 2003. Combining inferences from models of capture efficiency, detectability, and suitable habitat to classify landscapes for conservation of threatened bull trout. *Conservation Biology* 17: 1070-1077.
- Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. *BioScience* 47:769–784.
- Pollock, K. H., J. D. Nichols, T.R. Simons, G.L. Farnsworth, L.L. Bailey, and J. R. Sauer 2002. Large scale wildlife monitoring studies statistical methods for design and analysis. *Evironmetrics* 13: 105-119.
- Rabeni, C.F. 1985. Resource partitioning by stream dwelling crayfish: the influence of body size. *American Midland Naturalist* 113(1):20-29.
- Rabeni, C.F., M. Gosset, and D.D. McClendon. 1995. Contribution of crayfish to benthic invertebrate production and trophic ecology of an Ozark stream. *Freshwater Crayfish* 10:163-173
- Rabeni, C.F. , K. J. Collier, S. M. Parkyn, and B. J. Hicks. 1997. Evaluating techniques for sampling stream crayfish (*Paranephrops planifrons*).—*New Zealand Journal of Marine and Freshwater Research* 31: 693–700.

- Ratcliffe, J. A. and D. R. DeVries. 2004. The crayfishes (Crustacea: Decapoda) of the Tallapoosa River Drainage, Alabama. *Southeastern Naturalist* 3:417-430.
- Royle, J.A., and J.D. Nichols 2003. Estimating abundance from repeated presence-absence data or point counts. *Ecology* 84 (3):777-790.
- Royle, J.A., J.D. Nichols, and M. Kery. 2005. Modeling occurrence and abundance of species when detection is imperfect. *Oikos* 110:353-359.
- Schuster, G.A., C.A Taylor, and J. Johansen. 2008. An Annotated Checklist and Preliminary Designation of Drainage Distributions of the Crayfishes of Alabama. *Southeastern Naturalist* 7(3):493-504.
- Stein, R.A., and J.J. Magnuson 1976. Behavioral response of crayfish to a fish predator. *Ecology*. 57(4): 751-761.
- Taylor, R.C. 1988. Population Dynamics of the crayfish *Procambarus spiculifer* observed in different sized-streams in response to drought. *Journal of Crustacean Biology* 8(3): 40-409
- Taylor, C.A. 2002. Taxonomy and conservation of native crayfish stocks. In: Holdich D.M. editor. *Biology of freshwater crayfish*. Blackwell Sciences: Oxford pg. 236-257.
- Taylor, C. A. and G. A. Schuster. *The Crayfishes of Kentucky*. 2004. Illinois Natural History Survey Special Publication 28
- Taylor C. A., G. A. Schuster, J. E. Cooper, R. J. DiStefano, A. G. Eversole, P. Hamr, H. H. Hobbs III, H. W. Robison, C. E. Skelton, and R. F. Thoma 2007. Reassessment of the conservation status of crayfishes of the United States and Canada after 10+ years of increased awareness. *Fisheries* 32(8):372-389.
- Travnichek, V. H., M. B. Bain, and M. J. Maceina. 1995. Recovery of a warmwater fish assemblage after the initiation of a minimum-flow release downstream from a hydroelectric dam. *Transactions of the American Fisheries Society* 124: 836–844.
- Walters, C. J. 1986. *Adaptive management of renewable resources*. Macmillan Publishing, New York, New York.
- Werner, E.E. and J.F. Gilliam. 1984. The ontogenetic niche and species interactions in size structured populations.
- Whitledge G.W., and C.F. Rabeni. 1997. Energy sources and ecological role of crayfishes in an Ozark stream: insight from stable isotope gut content analysis. *Canadian Journal of Fisheries and Aquatic Science* 54:2555-2563.

Yoccoz, N. G., J. D. Nichols, and T. Boulinier. 2001. Monitoring of biological diversity in space and time; concepts, methods and designs. *Trends in Ecology and Evolution* 16: 446-453.

Appendix 1. Sampling dates for crayfish collection in the Tallapoosa River basin by site and year. Sites are arranged in a longitudinal fashion descending from R. L. Harris Dam. Sites below the solid black line are unregulated sites (Hillabee Creek between Sanford Road and Alabama Hwy 22 and the Upper Tallapoosa River between Ben Mills and Evans Ferry. Dashes indicate no collections made due to site inaccessibility.

	2005		2006		2007		2008		2009	
Dam to Malone A	24-Jun	13-Oct	26-May	18-Oct	6-Jun	26-Sep	2-Jul	10-Oct	14-May	16-Sep
Dam to Malone B	27-Jun	12-Oct	25-May	20-Oct	6-Jun	28-Sep	1-Jul	1-Oct	13-May	11-Sep
Dam to Malone C	27-Jun	12-Oct	25-May	20-Oct	6-Jun	28-Sep	1-Jul	1-Oct	13-May	11-Sep
Dam to Malone D	27-Jun	12-Oct	25-May	30-Oct	6-Jun	28-Sep	1-Jul	1-Oct	13-May	11-Sep
Dam to Malone E	27-Jun	12-Oct	25-May	30-Oct	7-Jun	28-Sep	1-Jul	1-Oct	13-May	11-Sep
Malone to Wadley A	20-Jun	20-Oct	31-May	6-Oct	20-Jun	19-Sep	6-Aug	17-Oct	1-Jun	29-Sep
Malone to Wadley B	20-Jun	20-Oct	31-May	6-Oct	20-Jun	19-Sep	6-Aug	17-Oct	1-Jun	29-Sep
Malone to Wadley C	21-Jun	19-Oct	1-Jun	13-Oct	13-Jun	19-Sep	6-Aug	10-Oct	1-Jun	2-Oct
Malone to Wadley D	21-Jun	19-Oct	1-Jun	13-Oct	13-Jun	21-Sep	7-Aug	13-Oct	2-Jun	2-Oct
Malone to Wadley E	21-Jun	19-Oct	1-Jun	13-Oct	13-Jun	21-Sep	7-Aug	13-Oct	2-Jun	2-Oct
Horseshoe Bend A		17-Nov	-	6-Nov	-	26-Sep	24-Jun	22-Sep	12-Jun	-
Horseshoe Bend B	-	17-Nov	-	6-Nov	-	26-Sep	24-Jun	22-Sep	12-Jun	-
Horseshoe Bend C	-	17-Nov	-	6-Nov	-	3-Oct	24-Jun	22-Sep	12-Jun	-
Horseshoe Bend D	-	15-Nov	-	23-Oct	-	3-Oct	19-Jun	26-Sep	20-Jul	-
Horseshoe Bend E	-	15-Nov	-	23-Oct	-	3-Oct	19-Jun	26-Sep	20-Jul	-
Hillabee Creek A	-	5-Oct	6-Jun	20-Sep	7-Aug	17-Oct	18-Jun	15-Sep	25-Jun	-
Hillabee Creek B	-	5-Oct	7-Jun	27-Sep	8-Aug	17-Oct	18-Jun	17-Sep	24-Jun	18-Sep
Hillabee Creek C	-	5-Oct	7-Jun	27-Sep	-	-	18-Jun	17-Sep	24-Jun	18-Sep
Hillabee Creek D	-	5-Oct	7-Jun	27-Sep	-	-	18-Jun	17-Sep	24-Jun	18-Sep
Hillabee Creek E	-	13-Oct	6-Jun	22-Sep	7-Aug	17-Oct	19-Jun	15-Sep	25-Jun	-
Upper Tallapoosa A	-	30-Sep	13-Jun	13-Nov	6-Aug	-	-	-	22-Jun	4-Sep
Upper Tallapoosa B	-	30-Sep	13-Jun	13-Nov	6-Aug	-	-	-	22-Jun	4-Sep
Upper Tallapoosa C	-	30-Sep	13-Jun	13-Nov	6-Aug	-	-	-	22-Jun	4-Sep
Upper Tallapoosa D	-	28-Sep	14-Jun	13-Nov	24-Jul	24-Oct	8-Jul	6-Oct	22-Jun	4-Sep
Upper Tallapoosa E	-	28-Sep	14-Jun	13-Nov	24-Jul	24-Oct	8-Jul	6-Oct	22-Jun	4-Sep

Appendix 2.- Summary of detections for *P. spiculifer* and number of samples taken in the Tallapoosa River basin by site and year. Sites are arranged in a longitudinal fashion descending from R.L. Harris Dam. Sites below the solid black line are unregulated sites (Hillabee Creek between Sanford Road and Alabama Hwy 22 and the Upper Tallapoosa River between Ben Mills and Evans Ferry. Dashes indicate no collections made due to site inaccessibility.

	2005	Samples	2006	Samples	2007	Samples	2008	Samples	2009	Samples
Dam to Malone A	4	40	4	40	7	40	4	10	2	20
Dam to Malone B	6	40	4	40	7	40	6	10	10	20
Dam to Malone C	7	40	8	40	11	40	3	10	9	20
Dam to Malone D	4	40	8	38	3	40	0	10	7	20
Dam to Malone E	5	40	6	40	10	40	0	10	10	20
Malone to Wadley A	4	40	9	40	1	40	2	10	7	20
Malone to Wadley B	1	40	3	40	1	40	4	10	2	20
Malone to Wadley C	3	40	6	40	8	40	2	10	7	20
Malone to Wadley D	5	40	5	40	3	40	0	10	1	20
Malone to Wadley E	7	40	2	40	7	40	1	10	7	20
Horseshoe Bend A	2	20	2	20	1	20	0	10	0	10
Horseshoe Bend B	1	20	3	20	0	20	0	10	1	10
Horseshoe Bend C	4	20	2	20	0	20	0	10	0	10
Horseshoe Bend D	5	20	0	20	0	20	0	10	0	10
Horseshoe Bend E	3	20	0	20	0	20	0	10	0	10
Hillabee Creek A	1	20	9	40	4	17	0	10	2	10
Hillabee Creek B	0	20	1	40	1	17	0	10	0	10
Hillabee Creek C	9	20	10	40	-	0	0	10	1	20
Hillabee Creek D	4	20	13	40	-	0	0	10	0	20
Hillabee Creek E	7	20	12	40	0	16	0	10	0	10
Upper Tallapoosa A	4	20	7	40	3	20	-	10	2	20
Upper Tallapoosa B	1	20	9	40	3	20	-	10	3	20
Upper Tallapoosa C	2	20	14	40	5	20	-	10	5	20
Upper Tallapoosa D	2	20	5	40	3	30	3	10	1	20
Upper Tallapoosa E	1	20	4	40	4	25	3	10	3	20

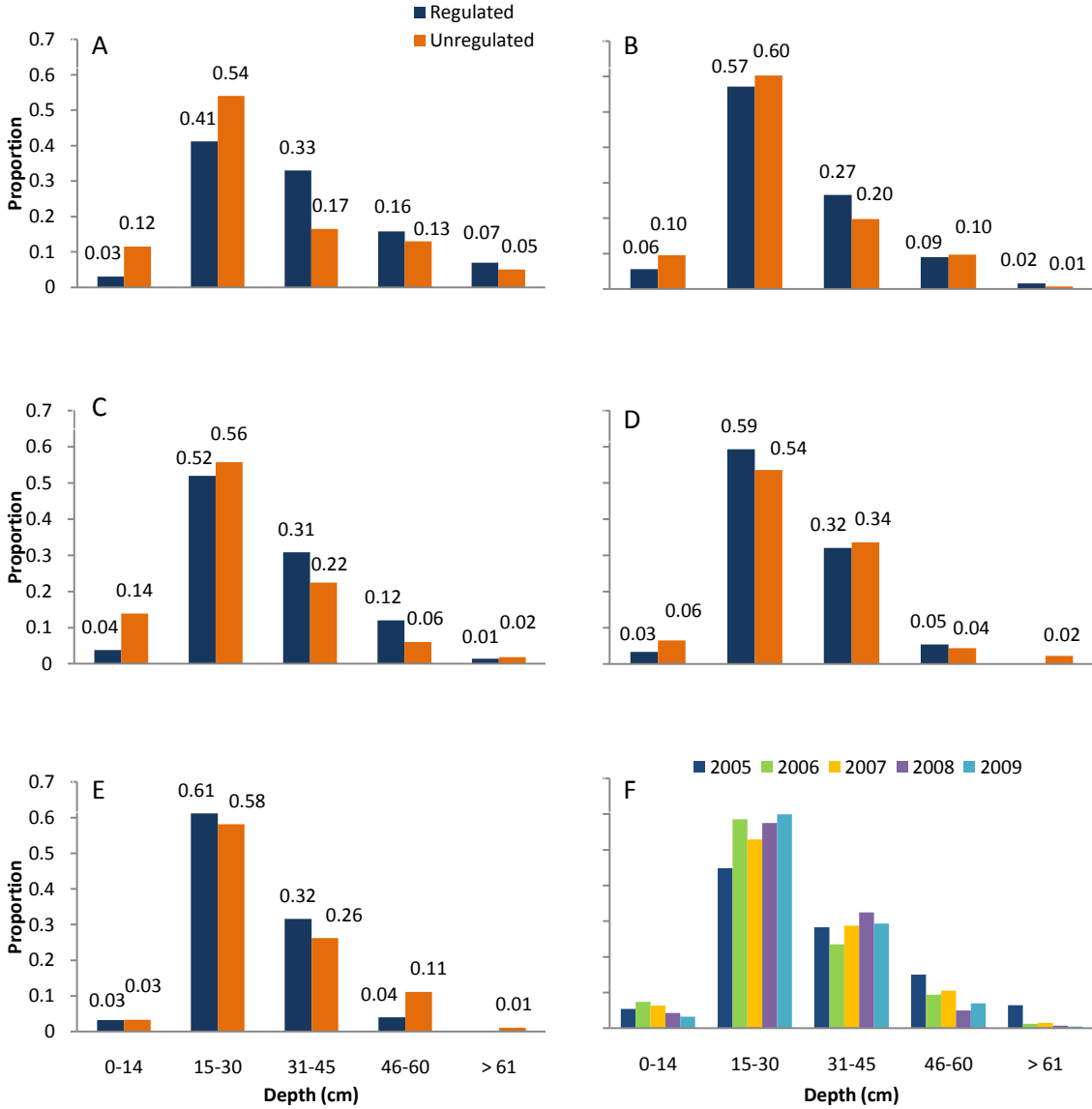
Appendix 3.- Summary of detections for *C. englishi* and number of samples taken in the Tallapoosa River basin by site and year. Sites are arranged in a longitudinal fashion descending from R. L. Harris Dam. Sites below the solid black line are unregulated sites (Hillabee Creek between Sanford Road and Alabama Hwy 22 and the Upper Tallapoosa River between Ben Mills and Evans Ferry. Dashes indicate no collections made due to site inaccessibility.

	2005	Samples	2006	Samples	2007	Samples	2008	Samples	2009	Samples
Dam to Malone A	0	40	0	40	1	40	0	10	1	20
Dam to Malone B	6	40	5	40	1	40	6	10	11	20
Dam to Malone C	3	40	4	40	9	40	3	10	4	20
Dam to Malone D	1	40	5	38	3	40	0	10	4	20
Dam to Malone E	12	40	3	40	7	40	2	10	2	20
Malone to Wadley A	5	40	3	40	0	40	1	10	4	20
Malone to Wadley B	2	40	2	40	4	40	2	10	5	20
Malone to Wadley C	3	40	14	40	7	40	4	10	6	20
Malone to Wadley D	3	40	7	40	1	40	0	10	5	20
Malone to Wadley E	0	40	2	40	0	40	0	10	2	20
Horseshoe Bend A	0	20	0	20	2	20	0	10	2	10
Horseshoe Bend B	0	20	1	20	0	20	0	10	0	10
Horseshoe Bend C	1	20	3	20	0	20	2	10	2	10
Horseshoe Bend D	2	20	0	20	0	20	0	10	0	10
Horseshoe Bend E	0	20	1	20	1	20	0	10	0	10
Hillabee Creek A	0	20	0	40	0	17	1	10	0	10
Hillabee Creek B	0	20	0	40	0	17	0	10	0	10
Hillabee Creek C	1	20	1	40	-	0	0	10	1	20
Hillabee Creek D	1	20	0	40	-	0	1	10	0	20
Hillabee Creek E	0	20	0	40	0	16	0	10	0	10
Upper Tallapoosa A	0	20	1	40	0	20	-	10	2	20
Upper Tallapoosa B	0	20	1	40	0	20	-	10	0	20
Upper Tallapoosa C	2	20	3	40	2	20	-	10	2	20
Upper Tallapoosa D	2	20	0	40	3	30	3	10	2	20
Upper Tallapoosa E	0	20	0	40	0	25	2	10	0	20

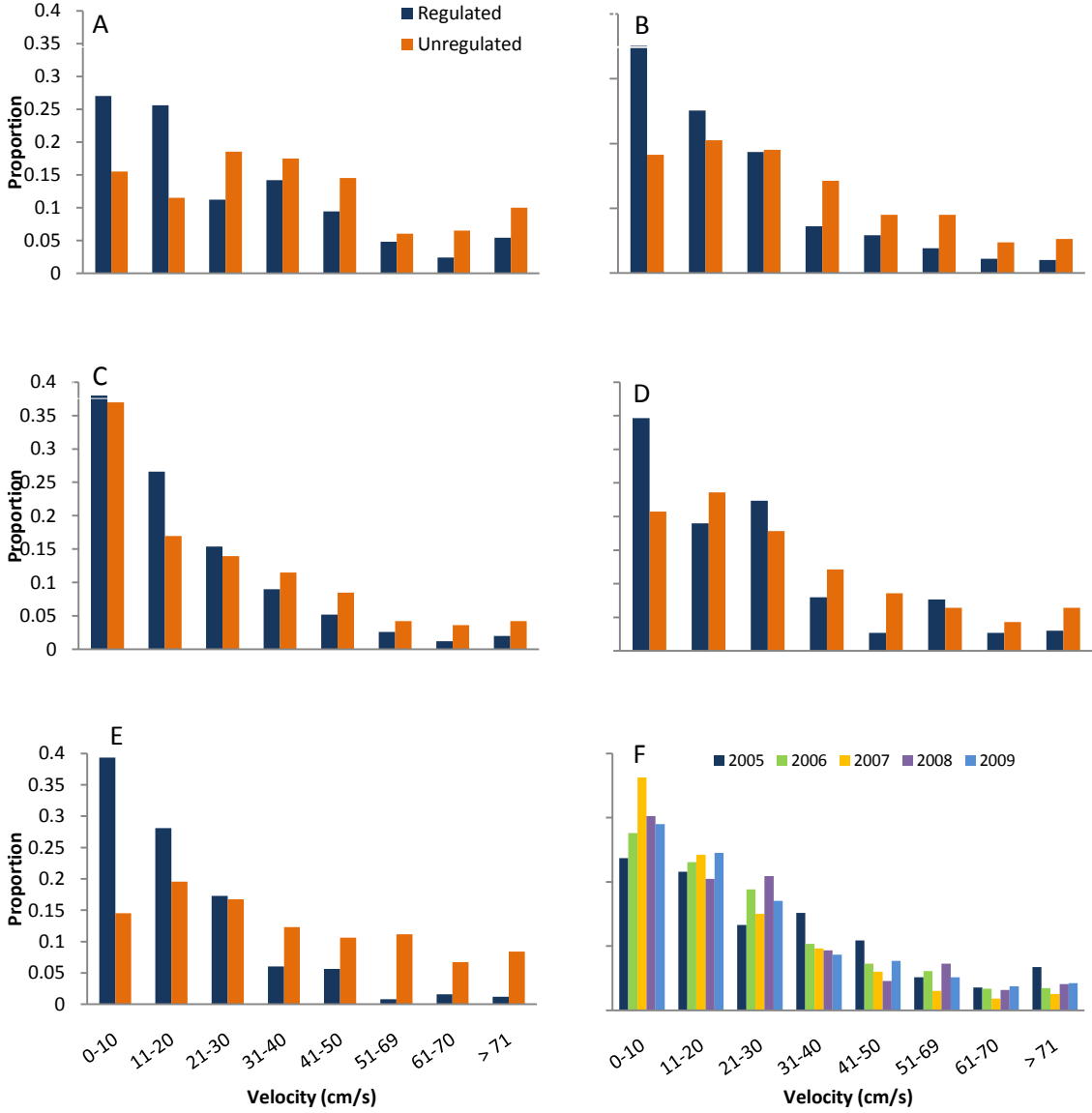
Appendix 4.- Summary of detections for *C. halli* and number of samples taken in the Tallapoosa River basin by site and year. Sites are arranged in a longitudinal fashion descending from R. L. Harris Dam. Sites below the solid black line are unregulated sites (Hillabee Creek between Sanford Road and Alabama Hwy 22 and the Upper Tallapoosa River between Ben Mills and Evans Ferry. Dashes indicate no collections made due to site inaccessibility.

	2005	Samples	2006	Samples	2007	Samples	2008	Samples	2009	Samples
Dam to Malone A	0	40	0	40	0	40	0	10	1	20
Dam to Malone B	1	40	1	40	2	40	2	10	0	20
Dam to Malone C	1	40	2	40	1	40	0	10	2	20
Dam to Malone D	1	40	3	38	1	40	0	10	1	20
Dam to Malone E	2	40	2	40	1	40	0	10	1	20
Malone to Wadley A	0	40	0	40	0	40	0	10	0	20
Malone to Wadley B	0	40	0	40	1	40	0	10	2	20
Malone to Wadley C	1	40	4	40	8	40	0	10	2	20
Malone to Wadley D	0	40	3	40	1	40	0	10	1	20
Malone to Wadley E	0	40	1	40	3	40	0	10	0	20
Horseshoe Bend A	1	20	4	20	0	20	2	10	0	10
Horseshoe Bend B	1	20	4	20	0	20	0	10	0	10
Horseshoe Bend C	2	20	0	20	0	20	1	10	0	10
Horseshoe Bend D	0	20	0	20	0	20	0	10	0	10
Horseshoe Bend E	0	20	0	20	0	20	0	10	0	10
Hillabee Creek A	1	20	1	40	0	17	0	10	0	10
Hillabee Creek B	0	20	0	40	0	17	0	10	0	10
Hillabee Creek C	2	20	0	40	-	0	0	10	0	20
Hillabee Creek D	1	20	1	40	-	0	0	10	0	20
Hillabee Creek E	1	20	3	40	1	16	0	10	0	10
Upper Tallapoosa A	0	20	1	40	5	20	-	10	0	20
Upper Tallapoosa B	1	20	4	40	1	20	-	10	1	20
Upper Tallapoosa C	2	20	5	40	4	20	-	10	5	20
Upper Tallapoosa D	1	20	4	40	6	30	2	10	1	20
Upper Tallapoosa E	0	20	3	40	1	25	1	10	0	20

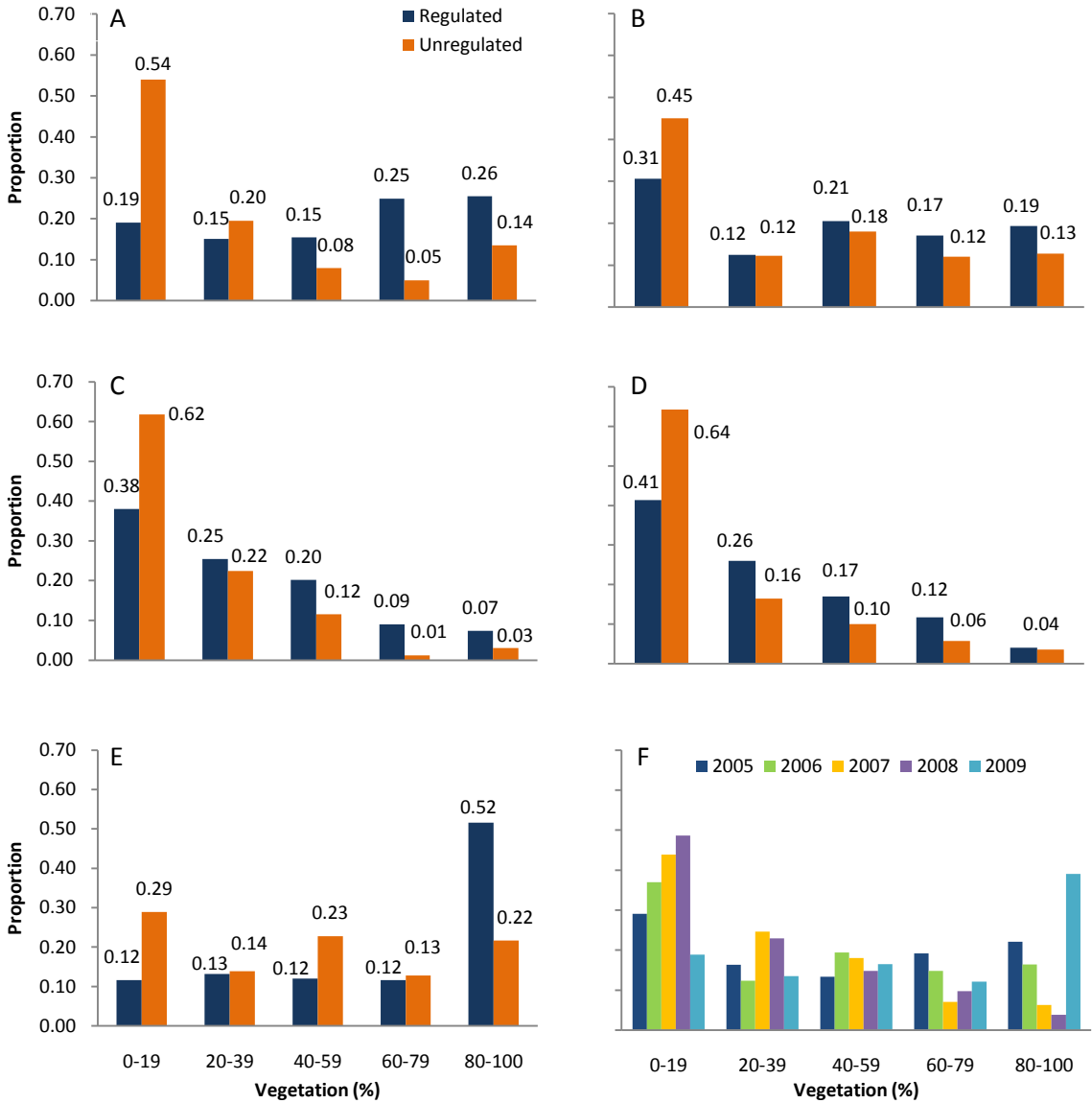
Appendix 5.- Distribution of depths in sampling units by proportion of samples taken in regulated versus unregulated reaches by year (A) 2005, (B) 2006, (C) 2007, (D) 2008, (E) 2009 and (F) distribution of depths in all sampling units in Tallapoosa River basin by year.



Appendix 6.- Distribution of velocities in sampling units by proportion of samples taken in regulated versus unregulated reaches (A) 2005, (B) 2006, (C) 2007, (D) 2008, (E) 2009 and (F) distribution of velocities in all sampling units in Tallapoosa River basin by year.



Appendix 7.- Distribution of vegetation (% areal coverage) in sampling units by proportion of samples taken in regulated versus unregulated reaches (A) 2005, (B) 2006, (C) 2007, (D) 2008, (E) 2009 and (F) distribution of percent vegetative cover in all sampling units in Tallapoosa River basin by year.



Appendix 8.- Distribution of substrate types in sampling units by proportion of samples taken in regulated versus unregulated reaches of the Tallapoosa River basin 2005-2009. Data represents the largest substrate size category present in the sampling unit.

